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CRITICAL REVIEW AND SUMMARY OF LEACHATE AND GAS PRODUCTION FROM LANDFILLS

by

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16. ABSTRACT

A Cooperative Agreement Between the Municipal Environmental Research Laboratory and the Georgia Institute of Technology was established in 1983 to provide an evaluation of the state-of-the-art in municipal waste, landfill leachate and gas management. Accordingly, summaries of full-scale and experimental-scale data on leachate and gas characteristics, control methods, and the performance of a number of biological and physical-chemical treatment alternatives have been developed and are presented together with recommendations for process implementation and future research.

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FOREWORD

The Environmental Protection Agency was created because of increasing public and governmental concern about the nation's environment and its effect on the health and welfare of the American people. The complexity of the environment and the interplay among its components require a concentrated and integrated attack upon environmental problems.

The first step in seeking environmental solution is research and development to define the problem, measure its impact and project possible remedies. Research and development is carried out continually by both industry and governmental agencies concerned with improving the environment. Much key research and development is conducted by EPA's Hazardous Waste Engineering Research Laboratory. The laboratory develops new and improved technologies and systems to treat, store and dispose hazardous waste; to remove hazardous waste; to remove hazardous waste and restore contaminated sites to usefulness; and, to promote waste reduction and recycling. This publication is one of the products of that research -- a vital communications link between the research and the user community.

This document presents a critical review and summary of available information and literature on leachate and gas production and management during the disposal of solid wastes.

Thomas Hauser, Director
Hazardous Waste Engineering Research Laboratory

ABSTRACT

The purpose of this project was to provide a review and critique of literature and the status of technology pertaining to the formation, characterization and treatment of municipal solid waste (MSW) landfill gas and leachate. This objective was accommodated by an initial recognition of the various factors influencing the applicability and efficacy of various landfill disposal options and/or leachate and gas management strategies. Therefore, the review is organized to reflect bench-, pilot- and full-scale evaluations; leachates have been categorized according to low, medium and high strengths; and product gases have been differentiated on the basis of constituents and medium or high energy content.

Commencing with a brief historical perspective focusing on the problems associated with migration of MSW leachate and gas from landfill disposal sites, the review proceeds with a characterization of MSW gas and leachate quality produced throughout the phases of landfill stabilization and the various leachate and gas treatment methods. Accordingly, leachate treatment by external aerobic and anaerobic biological methods and by in situ anaerobic degradation with leachate management are compared to physical-chemical leachate treatment by chemical oxidation, precipitation, coagulation, disinfection, absorption, ion exchange and reverse osmosis. In addition, methods of landfill gas collection and treatment by sorption, refrigeration, and membrane separation are presented together with the implications of ultimate disposal of treated leachates, including an understanding of leachate/soil interactions.

Biological treatment is shown to be superior to physical-chemical methods for raw leachate; aerobic and anaerobic treatment methods were generally capable of removing 85 to 99 percent of leachate BOD_5 and COD as well as most heavy metals. The physical-chemical processes are shown to be better suited for polishing of biological treatment effluents and for the removal of refractory organics. Leachate management through recycle is also demonstrated to possess a high potential for process stability, although further effort is needed to determine application criteria on a large scale. Process loadings, operational constraints, and degree of variance and uncertainty in reported data and their interpretation are presented to indicate that past efforts have not resolved all the technical challenges associated with gas and leachate management at landfill disposal sites, and that additional investigations are required.

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SECTION 1

INTRODUCTION

Sanitary landfills continue to be the most frequently employed method of solid waste disposal practiced in the United States. Unfortunately, sanitary landfills remain poorly understood and loosely managed; deficiencies magnified and manifested by usual inadequacies in waste definition and understanding of associated environmental variables. During the last decade, the potential for production of leachate and gas has received major attention particularly in terms of environmental consequences associated with the migration of leachate and gas during conversion of waste constituents. These concerns have led to a variety of developments for control, including the concepts of leachate containment and total landfill isolation. In accordance with these strategies, various techniques have been proposed and implemented for the treatment and disposal of landfill gases and leachates.

The purpose of this report is to provide a review and summary of the nature of leachate and gas production at landfills, and to couple this with a concomitant inventory of available techniques for containment, control and treatment. The review begins with a brief historical perspective of hazards associated with the migration of leachate and gas from landfill disposal sites. Factors affecting the quantity and quality of landfill leachate and gas are then addressed, followed by processes used or advocated for leachate and gas treatment. Hence, investigations into activated sludge, aerated lagoons, trickling filters, biodisks, anaerobic contact processes and *in situ* leachate recycle technologies as well as coagulation, precipitation, chemical oxidation, disinfection, adsorption, ion exchange, and reverse osmosis processes in either separate or combined configurations are detailed. Finally, methods for the ultimate disposal of leachate and gas are addressed, including discharge to municipal wastewater treatment plants, land application, and energy recovery.

SECTION 2

CONCLUSIONS

GENERAL

The development of rational and economically sound solutions to landfill leachate and gas migration hazards encompasses the analysis of several major factors. As shown in Figure 1, a given landfill in its natural setting will affect and be affected by numerous hydrologic and geologic circumstances that must be properly recognized and managed to minimize human and environmental risks. In particular, leachate and gases formed as a consequence of external moisture inputs and waste degradation may migrate into the surrounding environment, contaminate drinking water supplies and create other environmental hazards.

The first step towards effective management of gas and leachates at susceptible landfill sites logically begins with containment by installation of "impermeable" barriers augmented by drainage, venting, and collection systems sufficient to handle the inevitable production of leachate and gas. Following their generation and capture, leachate and gas must be treated and disposed of in an environmentally acceptable and economically sound manner.

As also shown in Figure 1, there are a number of options available for leachate and gas management prior to ultimate disposal. Before being discharged onto land or into a publicly owned treatment works (POTW), landfill leachate and gas will require treatment by biological and/or physical-chemical methods. Some of these methods have been proven successful, while others have been shown to have limited applicability. Moreover, it is widely recognized that the quantity and quality of landfill leachate and gas are influenced by numerous variables which have resulted in a diversity of relative treatment efficiences when similar processes have been applied. However, some generalizations on the advantages and disadvantages of these processes have become evident, as are outlined in the remainder of this section.

LEACHATE TREATMENT PROCESS PERFORMANCE

When considering separate treatment of raw leachate for removal of biodegradable fractions, biological treatment systems were significantly superior to physical-chemical techniques. As indicated in the performance summary presented in Table 1, if given sufficient residence time, biological processes typically achieved 90 to 99% organics (BOD_5 and COD) removal and yielded effluents having COD concentrations less than 500 mg/l. The aerobic treatment processes were generally capable of 90% NH_3-N conversion and typically yielded effluents containing less than 10 mg/l NH_3-N for θ_c >10 days.

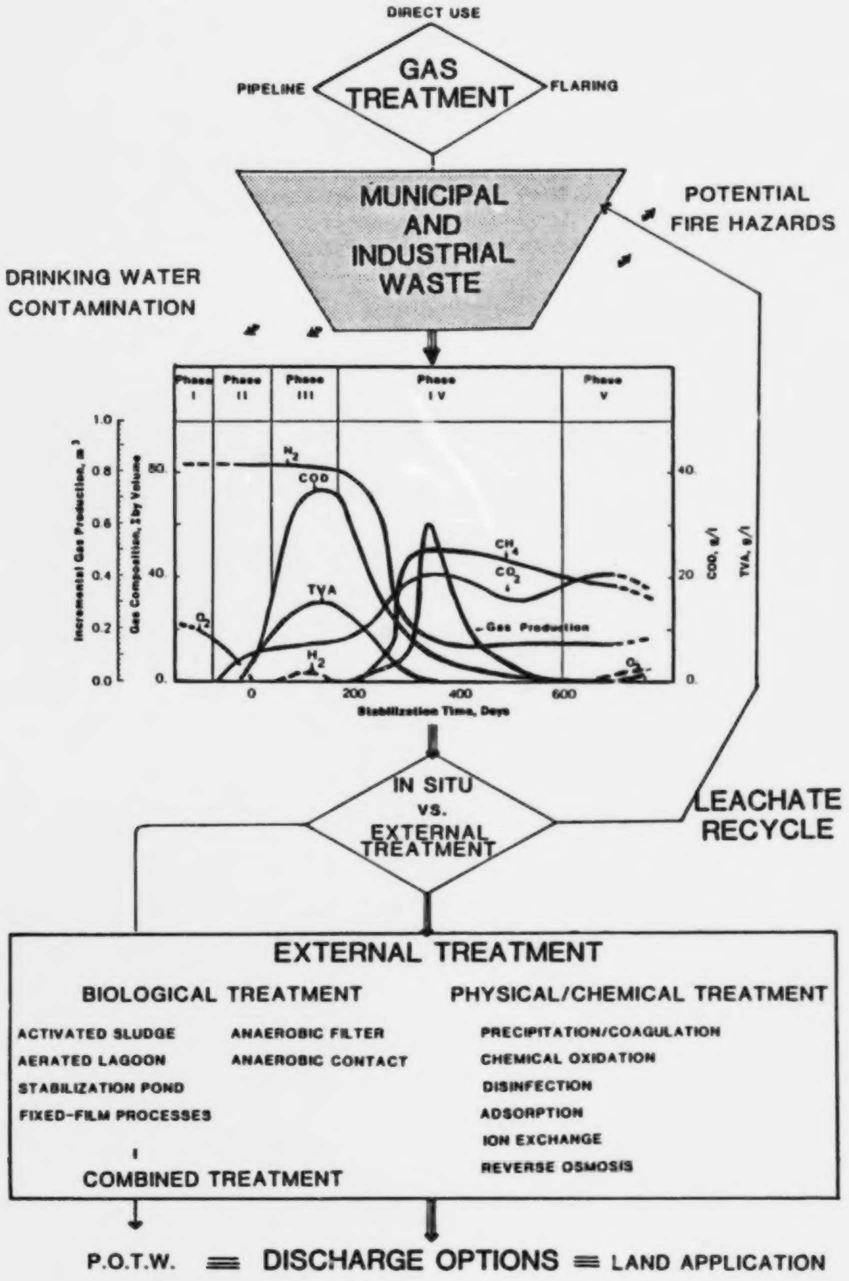
DIRECT USE
PIPELINE
GAS TREATMENT
FLARING
MUNICIPAL AND INDUSTRIAL WASTE
POTENTIAL FIRE HAZARDS
DRINKING WATER CONTAMINATION
Phase I
Phase II
Phase III
Phase IV
Phase V
Incremental Gas Production, m³
Gas Composition, % by Volume
COD, g/l
TVA, g/l
H2
COD
CH4
CO2
TVA
O2
Gas Production
0
200
400
600
Stabilization Time, Days
IN SITU vs. EXTERNAL TREATMENT
LEACHATE RECYCLE
EXTERNAL TREATMENT
BIOLOGICAL TREATMENT
PHYSICAL/CHEMICAL TREATMENT
ACTIVATED SLUDGE
AERATED LAGOON
STABILIZATION POND
FIXED-FILM PROCESSES
ANAEROBIC FILTER
ANAEROBIC CONTACT
PRECIPITATION/COAGULATION
CHEMICAL OXIDATION
DISINFECTION
ADSORPTION
ION EXCHANGE
REVERSE OSMOSIS
COMBINED TREATMENT
P.O.T.W. ≡ DISCHARGE OPTIONS ≡ LAND APPLICATION

Figure 1. Treatment Options Available for Leachate and Gas Management and Ultimate Disposal

TABLE 1. SUMMARY OF LEACHATE TREATMENT PROCESS CAPABILITIES

	BOD_5		COD		TKN		Fe		Zn		Ni		Comments
	Rem., %	Effl., mg/l	Rem., %	Effl., mg/l	Rem., %	Effl., mg/l	Rem., %	Effl., mg/l	Rem., %	Effl., mg/l	Rem., %	Effl., mg/l	
AEROBIC BIOLOGICAL PROCESSES													
Activated Sludge	95	100	95	500	70-95	10-100	96-99	10-40	96-99	3-10	60	0.25	θ_c = 6-10 days
Combined Leachate and Sewage	94-99	3-15	92-98	25-60	--	--	--	--	--	--	--	--	ratio <5%
Aerated Lagoon	99	5-60	92-98	300-800	40-70*	40-80	99	0.2	--	--	--	--	θ_c >10 days
Stabilization Pond	93-99	10-100	99	100-400	70-99	4-100	80-99	1-100	--	--	--	--	τ >40 days
Aerobic Fixed Film*													
ANAEROBIC BIOLOGICAL PROCESSES													
Attached Growth	85-98	100-900	75-95	200-1000	--	--	80-99	5-25	80-99	0.5-10	10-80	0.1-1	θ_c >10 days
Suspended Growth	85-98	100-900	75-95	200-1000	--	--	80-99	5-25	80-99	0.5-10	10-80	0.1-1	θ_c >5 days
Leachate Recycle	NA	<100	NA	<5	NA	20-1000	NA	5-50	NA	0.2-1	NA	--	θ_c >500 days
PHYSICAL/CHEMICAL PROCESSES													
Coagulation	--	--	12	100-10,000	--	--	95-99	2-17	75-98	<1	--	--	Lime, alum, ferric chloride
Oxidation	--	--	10-50				99	<1	90	<1	--	--	Ozone, chloride permanganate
Reverse Osmosis	--	--	60-90** 86-94	1000-8000 <10	--	--	--	--	--	--	--	--	Raw Leachate Pretreated Leachate
Ion Exchange	--	--	40-70	100-300	--	--	40-80	1-10	20-96	<1	14-96	<1	Commercial IX Resins and GG
Adsorption	--	--	75-99	<10	--	--	65-95	2-15	--	--	--	--	GAC and PAC

Rem. = Removal; Effl. = Effluent

*Insufficient data to make an adequate judgment; **TOC Basis

For θ_c of 6 to 10 days, the limiting range for aerobic carbonaceous material conversion, 60 to 80% nitrification was generally also achieved.

Like the aerobic biological processes, anaerobic biological processes have been successfully applied for treatment of raw leachates. COD and BOD_5 removals of 90% were typically achieved at residence times longer than 10 days. With these conditions, gas production from anaerobic processes ranged from 0.4 to 0.6 m^3/kg COD destroyed or 0.8 to 0.9 m^3/kg BOD_5 destroyed.

Aerobic biological processes were fairly efficiently applied for removal of heavy metals. Zinc, iron, cadmium and manganese were removed best, followed by lower removals of chromium, lead and nickel. Zinc, chromium, and iron were removed at efficiencies greater than 90% during anaerobic treatment; copper, lead, cadmium, and nickel removals were on the order of 50 to 90%. Removals of alkaline earth metals were relatively unaffected in both aerobic and anaerobic processes, although the activated sludge process has been reported to remove 64 to 99% calcium.

With the exception of activated carbon, the physical-chemical processes were generally unsuccessfully applied for removal of organic materials from raw leachates. However, reverse osmosis, activated carbon (GAC and PAC) and ion exhange (IX) were successfully applied to treated effluents from biological treatment processes. Reverse osmosis treatment removed a high percentage of organics from both raw and treated leachates, although fouling problems limited its applicability to raw leachates. Ion exchange treatment was generally ineffective for organics removal, although cation exchange resins such as glauconitic greensand (GC) were successful in removing copper, lead and nickel (these were poorly removed in biological processes). Iron and zinc were also relatively well removed, as were chromium, manganese, calcium and magnesium.

Activated carbon adsorption was shown to be capable of removing the majority of residual organics from chemical and biological leachate treatment process effluents, yielding BOD_5 concentrations after adsorption of less than 50 mg/l. Raw leachates have also been treated using activated carbon, achieving >95% TOC removal (<100 mg/l effluent) with a maximum adsorptive capacity of 200 mg TOC/g AC.

In situ treatment of leachate using leachate containment and recycle back through the landfill mass has been demonstrated to be successful on pilot- and full-scale. Effluents from leachate recycle studies were typically 30 to 350 mg/l BOD_5, 70 to 500 mg/l COD, 4 to 40 mg/l iron and <1 mg/l zinc. The implementation of leachate recycle also generally reduced the time required for biological stabilization of the readily biologically degradable leachate constituents by as much as an order of magnitude. Whereas, wastes in landfills without leachate recirculation may require 15 to 20 years to stabilize, leachate recycle may shorten this period to 2 to 3 years. Moreover, if removal and ultimate disposal of accumulated leachate are followed by appropriate capping and maintenance of closed landfill sections, the potential for long-term adverse environmental impacts will be greatly diminished by concomitant removal of refractory substances remaining in the stabilized leachate and also depriving the system of that liquid (leachate) transport medium. Therefore, although the ultimate reactivity or fate of refractory compounds within landfills have not been well established, leachate recycle appears to offer a management option that can help reduce this degree of uncertainty and provide a better basis for predicting ultimate behavior.

GAS TREATMENT PROCESS PERFORMANCE

Effective recovery of energy (methane) from landfills requires appropriate provisions for gas collection and treatment, preferably conceived prior to the initiation of landfill operations. These systems need to be sized according to expected gas rates and yields. Based upon experiences recorded in the literature, from 0.005 to 0.10 m^3 of total gas have been produced per kilogram of dry refuse placed. Most of the total gas is produced over a relatively short period during the "life" of a landfill; the majority of methane will be produced within a few years after the onset of rapid stabilization and methanogenesis. Accordingly, typical gas production rates reported in the literature have ranged from 0.001 to 0.008 m^3/kg of dry refuse/year. With recycle augmented stabilization, these rates may be increased due to the shortened period (months versus years) for accelerated conversion of the readily available biodegradable materials present in the refuse and leachate. The associated gas composition has ranged from 45 to 60% methane with the balance being primarily carbon dioxide with smaller amounts of hydrogen, oxygen, nitrogen and traces of other gases.

The choice of treatment technologies utilized for purifying recovered landfill gas has depended on the intended use of the product. For high BTU pipeline quality gas, treatment has traditionally included the removal of water, carbon dioxide, hydrogen sulfide, hydrocarbons and, on occasion, nitrogen. For on-site use applications, lesser degrees of treatment have commonly been required, including the removal of water and hydrogen sulfide, but not necessarily carbon dioxide, hydrocarbons and nitrogen.

Water removal may be best effected by adsorption or absorption; absorption with ethylene glycol at <20°F (<6.7°C) appears to be the method of choice. Non-methane hydrocarbons may be removed using carbon adsorption. Carbon dioxide may be removed by organic solvents, alkaline salt solutions, or alkanolamines which seem to be the most popular. Hydrogen sulfide may be removed along with CO_2 by the above methods, or selectively removed by particular absorbents or adsorbents. Many of the solvent processes exhibit a higher affinity for H_2S than for CO_2, therefore, these gases may be removed concurrently in most cases. Dry oxidation processes (such as iron sponges) are more specific for hydrogen sulfide, although the non-regenerative nature of the support materials (such as wood shavings) often poses a requirement for additonal recharging procedures. Nitrogen may be removed by liquefying the methane fraction of landfill gas, although this is energy intensive which underscores the need to avoid introducing air during extraction from the landfill.

SECTION 3

RECOMMENDATIONS

GENERAL

The generation and treatment of landfill and leachate gas are influenced by many factors, many of which are poorly understood and ineffectively controlled or managed. Moreover, it is likely that the current practice of codisposal of small quantities of toxic and hazardous industrial wastes with municipal refuse will present increasing management challenges as leachates and gases are generated. Collectively, these issues have been emphasized by the results of studies reviewed herein with respect to the variations in quantity and quality of leachates and gases produced in time and space within a given landfill setting. Associated uncertainties tend to stymie management efforts and, as a result, the design, construction and operation of external leachate treatment facilities have not been standardized. Likewise, efforts directed toward energy (methane) recovery have been limited because of the difficulties in predicting variations in gas quality and production, as well as securing justification for such an initiative within the user community.

To help alleviate such problems during design and operation of leachate and gas management systems, it is desirable to have as much control over the generation of leachate and gas as possible and to thereby transfer the process from the realm of uncertainty to that of predictability. This can only be accomplished if control over leachate constituents is exercised either through the pre-selection of waste source ingredients or by management of their rate of generation and transfer to the transport medium (leachate or gas). The latter approach appears to be a more logical choice in the case of municipal landfills; the former, perhaps coupled with the latter, would seem more attractive for industrial landfills.

Based upon an understanding of the processes effecting leachate characteristics, management of generation and transfer rates can be implemented by management of the moisture regime within the landfill. Without moisture, the transport medium will not exist and the conversions and interactions determining leachate (and gas) quality will be suppressed. Once under control, the availability of moisture can be used to advantage to accelerate processes producing leachable constituents, to carry the constituents from the waste mass, to dilute out inhibitory ingredients and/or refractory compounds, to add seed, nutrients or buffer capacity to augment biological activity, and to transport residuals for ultimate treatment or disposal.

Implicit in this management concept are requirements for containment and ultimate disposal. Current technology provides a sufficiency of techniques for containment with natural or fabricated liners which have become generally accepted. Ultimate disposal relates to the sensitivity of the eventual environmental receptor, whether it be the land or the water. However, under prevailing regulatory constraints and state-of-the-art technology, both require some degree of leachate pretreatment before ultimate disposal is acceptable. It is the premise here that such pretreatment can be best provided in engineered systems that have the resiliency to cope with changing leachate characteristics.

In situ Treatment of Leachates

For on-site applications, it is recommended that leachate recycle be recognized as affording the flexibility needed to successfully manage landfill leachates, both with respect to leachate quality and quantity and energy recovery. Associated design of leachate and gas collection and distribution systems should be standardized and coupled with management plans allowing sequenced operation of the landfill and reuse of appurtenances to minimize overall costs and maximize the benefits of such treatment. Current evidence suggesting lower costs of leachate recycle in contained sites as compared to either separate aerobic or anaerobic treatment systems should be confirmed. In addition, since with leachate recycle the landfill itself provides the treatment system, operational contingencies should be established in relation to the accelerated production of leachate constituents and their eventual conversion to gas.

Whether leachate values are attractive for recovery and/or reuse also relates to the type of treatment provided. At many conventional municipal landfills, gross uncertainties persist throughout operation and after closure of the site. Accordingly, gas and leachate production events are generally unpredictable and neither gas nor leachate may be efficiently recovered for controlled discharge. With leachate recycle and its inherent ability to accelerate waste and leachate conversion with concomitant methane production, gas collection and possible utilization becomes more viable and such an option should be investigated further, particularly on full-scale. Moreover, the degree of stabilization of the waste mass as compared to conventional landfill practice needs to be established with regard to residual leachate character and decisions on ultimate leachate disposal including foreclosure and postclosure requirements.

External Treatment of Leachates and Gas

In the case of external treatment of leachates, the most logical first step appears to be biological treatment. Stabilization ponds or aerated lagoons can be most cost effective if land area is readily available; if not, anaerobic treatment or aerobic activated sludge processes may be used. The choice between anaerobic and aerobic processes for leachate treatment is a difficult one, although the retention times needed in either case are similar. Therefore, the energy surplus associated with methane production and aerator elimination may favor anaerobic processes. Both processes require further site specific testing on pilot- and full-scale to determine these issues. In particular, these systems will require attention to the flexibility in design and operation necessary to meet the challenges imposed by the stochastic nature of leachates (and gas) in both quality and quantity.

Following external biological treatment (or in situ treatment, as above), the effluents will still contain significant organic and inorganic residual concentrations. Therefore, polishing treatment prior to disposal on land or into a POTW such as by activated carbon adsorption, ion exchange or reverse osmosis needs to be included in the overall study approach. Precipitation and coagulation processes should also be considered where justified. In all cases, gas management or recovery need to be an integral part of any investigative initiative.

Directions for Future Research

Based upon the observations gained from this review, the present state-of-the-art in landfill leachate and gas management appears to be comprised of the elements represented in Figure 2. From this figure, it is suggested that 90 to 95% of the organics and metals leached from landfill waste may be removed by biological processes such as leachate recycle or external aerobic and anaerobic treatment systems. However, the capabilities of these processes are not fully established; further study is needed in each area to develop meaningful economic and realistic process control comparisons of these alternatives. Evaluations of leachate treatment and the gas production possible from the use of leachate recycle on full-scale are particularly needed, as well as parallel evaluations of both aerobic and anaerobic fixed-film processes on pilot- and full-scale, respectively. The sequence approach to leachate recycle on full-scale needs development to establish the economic incentives associated with minimizing leachate distribution and gas collection appurtenances and maximizing gas/recovery utilization. In all biological treatment cases, the stochastic nature of leachate and gas production in both quantity and quality needs to be merged with design and operational procedures.

Activated carbon, ion exchange or reverse osmosis polishing of effluents from biological treatment processes need further confirmation on full-scale. Included in these analyses should be a characterization of organics and inorganics escaping treatment, and the potential for improving final polishing by chemical pretreatment or post-treatment. Coupled with this initiative should be more detailed analyses of the character and fate of the priority pollutants appearing throughout the various phases of landfill stabilization and/or in situ or separate treatment.

Finally, the present knowledge of chemical and physical processes involved in management of gas and leachate from landfills is not sufficiently detailed to support development of a unified approach to leachate and gas treatment and possible resource recovery. Although much is known about factors that influence composition and volume of leachate and gas from landfills, this information is quantitative only for well controlled systems that are operated at laboratory or pilot scale. Many of the processes are not fully understood and information from field-scale systems remains qualitative and highly variable. Consequently, it is recommended that research and/or demonstration studies should be directed toward improving quantitative knowledge of the fundamental processes operative in field-scale systems and integrating this information into a strategy for standardized management and control of all types of landfills.

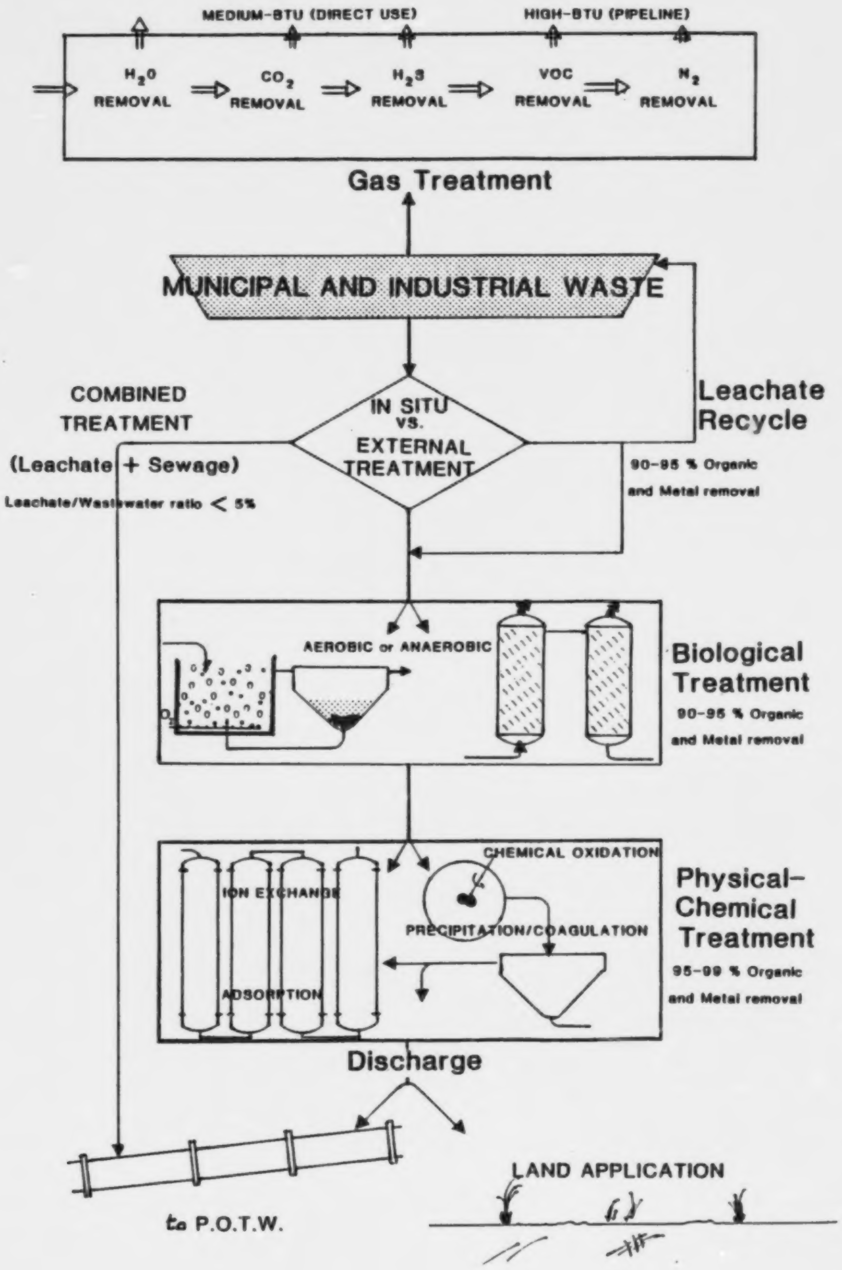
MEDIUM-BTU (DIRECT USE)
HIGH-BTU (PIPELINE)
H_2O REMOVAL
CO_2 REMOVAL
H_2S REMOVAL
VOC REMOVAL
N_2 REMOVAL
Gas Treatment
MUNICIPAL AND INDUSTRIAL WASTE
COMBINED TREATMENT
(Leachate + Sewage)
Leachate/Wastewater ratio < 5%
IN SITU vs. EXTERNAL TREATMENT
Leachate Recycle
90-95 % Organic and Metal removal
AEROBIC or ANAEROBIC
Biological Treatment
90-95 % Organic and Metal removal
CHEMICAL OXIDATION
ION EXCHANGE
PRECIPITATION/COAGULATION
ADSORPTION
Physical-Chemical Treatment
95-99 % Organic and Metal removal
Discharge
to P.O.T.W.
LAND APPLICATION

Figure 2. Solutions to the Management of Leachate and Gas from Landfill Disposal of Solid Wastes

SECTION 4

LANDFILL HAZARDS - HISTORICAL PERSPECTIVE

The technical literature has frequently documented problems associated with leachate and gas production from landfills, generally in terms of migration into the adjacent environment. Although often difficult to quantify, much of the earlier recorded information is instructive to the extent that it established a relatively early recognition and emphasis on these environmental problems. Therefore, a brief discussion of this early work is provided as an introduction to the more current investigations.

EARLY REPORTS ON LEACHATE MIGRATION AND EFFECTS

Potential problems associated with the burial of solid and liquid wastes have been documented as early as 1932 by Calvert (1932) who reported increased levels of hardness, calcium, magnesium, total solids, and carbon dioxide in a well more than 150 m from an impounding pit. Similarly, Carpenter and Setter (1940) sampled the water from the bottom of a refuse fill and reported contaminant concentrations of 1987 mg/l BOD_5, 3867 mg/l alkalinity as $CaCO_3$, and 3506 mg/l chloride. Lang (1941) reported the pollution of a well which was more than 600 m from a fill site. In a study on leaching of land-disposed wastes, Merz (1964) determined that if fill materials are allowed to contact groundwater either intermittently or continuously, the water becomes so grossly polluted to preclude its domestic or irrigation use.

Based on a study of an existing landfill located in an abandoned gravel pit, Anderson and Dornbush (1967) reported that groundwater in the immediate vicinity of the landfill, as well as that in direct contact with the landfill, exhibited an increase in ionic strength. Water quality impairment by excess ions decreased with distance from the landfill area. In studies on the characteristics of refuse tips in England, it was concluded that leaching could promote the growth of bacterial slimes and/or fungus in groundwater systems and lead to taste and odor problems (Davison, 1969). In California, groundwater below the Riverside Landfill contained markedly increased concentrations of BOD, chloride, sodium, and sulfate; increases of these contaminants over background were 26, 10, 9, and 8 times, respectively (Coe, 1970). Pollution of a surface water supply in Kansas City, MO was attributed to leaching of organic compounds from an industrial waste landfill into the Missouri River one mile (1.6 km) above the city's water intake (Hopkins and Poplisky, 1970). In 1975, the U.S. Environmental Protection Agency (EPA) assessed leachate damages from five municipal waste disposal sites reported by Fungaroli (1971) where contamination of groundwater and pollution of residential, industrial or public water supply wells occurred. The necessity to abandon all wells resulted in a costly replacement of the water supply.

Shuster (1976a) reported on the improper management of a landfill which resulted in serious contamination of public water supplies. Initially operated as an open dump, the City of Aurora, IL contracted in 1965 to have a private company operate the landfill at a site located over a creviced bedrock aquifer. Within two months of filling an excavated trench (dug to bedrock) with commerical, industrial, and septic tank wastes, seven residential wells were contaminated with leachate and declared totally unusable. BOD levels in three of the wells greatly exceeded levels reported for raw sewage. Shuster (1976b) similarly reported on a landfill site in Rockford, IL which was formerly a sand and gravel pit. Prior to placing the wastes into the site, sand was removed from a 161,880-m^2 (40-acre) area to a depth of 9.1 to 12.2 m (30-40 ft) below grade; the location of the water table was at 10.4 m (34 feet) below grade. As a result, four residential wells, four industrial wells and one public water supply well were contaminated. All wells were subsequently abandoned and alternative water supplies were established.

A 1977 Report to Congress presented data which documented contamination from various municipal and industrial land disposal sites (EPA, 1977). A total of 42 municipal and 18 industrial disposal sites were surveyed, and five of the municipal and 14 of the industrial sites were shown to contribute toxic pollutants to the local water supply. In all cases, groundwater contamination was the most common type of environmental damage.

EARLY REPORTS ON GAS MIGRATION AND EFFECTS

Landfill gases may also migrate from a landfill site and pose problems ranging from malodors and corrosion to fire or explosions. Methane migration and accumulation into subsurface structures, including sewerage system manholes and catch basins and into commercial and residential basements may explode if the methane is diluted with air to 5 to 15%. Zabetakis (1962) reported on the presence of methane at 2.1% in malodorous gases collected near water pipes at a number of homes built over a previous dump site. MacFarland (1970) summarized a report of a recreation center explosion in Atlanta, GA where two workmen were killed and two others seriously injured. As in the case of the homes, the recreation center had been rebuilt virtually on top of decomposing refuse. The explosion, which completely destroyed the recreation building, was attributed to methane orginating from the buried wastes. Flammable gas concentrations were found in nearly all probes placed within a 44.4-m (200-ft) radius of the demolished structure.

More recently, a British group (County Surveyors Society, Committee No. 4; 1982) surveyed 51 governmental agencies to determine the extent knowledge concerning landfill gas hazards. Of the agencies queried, 27 reported problems associated with fires and explosions. No deaths were reported associated with landfill explosion hazards, although two children were asphyxiated in a culvert extending from under a landfill in 1977. Injuries to children playing with matches or fireworks near manholes or drainage culverts were reported in several counties, and hazards to electrical, phone, and maintenance workers, including several large explosions, were also recorded. Numerous other incidents involving explosions and fires (62 total incidents) in which no personal injury was involved were also reported. Fires were apparently common both above and below ground, particularly in manholes.

In response to these problems, the control of landfill gas and leachate has received considerable attention and is often mandated by permit requirements. The current technology for gas control and treatment ranges from controlled ventilation to capture and processing for energy recovery. The planning and technological requirements associated with these approaches are addressed in more detail under the GAS MANAGEMENT section of this report. Similarly, the current technology for leachate management encompasses a variety of treatment or disposal options which are detailed in the LEACHATE TREATMENT section of this report.

SECTION 5

LEACHATE AND GAS PRODUCTION AT SANITARY LANDFILLS

GENERAL PERSPECTIVE

The chemical characterization of leachates and gases emanating from landfill operations is a first and necessary step toward a meaningful analysis of potential environmental impacts and the consideration of containment, control and treatment strategies. Unfortunately, the nature of landfill leachates (and gases to a lesser extent) varies widely in response to differences in climatic and hydrogeologic influences, the nature of the wastes contained at each site, and the age of the landfill (or its degree of stabilization). It is the purpose of this section to introduce and review the implications of these variables and to formulate an overall perspective and general characterization of landfill leachates and product gases.

CLIMATIC AND HYDROGEOLOGIC FACTORS

Rainfall provides the transport phase for leaching and migration of contaminants from a landfill and the moisture needed for biological activity leading to methane production. Although some moisture may be derived from the input wastes, the major precursor to leachate formation is infiltration from rainfall. The contact opportunity of this infiltration can be affected by certain landfill management options, including the cover configuration, liners, and the landfilling technique employed. In addition, the natural and imposed hydrogeologic conditions play a major role in determining the nature and fate of leachates and gases at sanitary landfills. Unfortunately, much uncertainty remains in this regard and analytical techniques often fail to reveal discontinuties in geologic and hydrologic dimensions.

Under similar infiltration constraints, the impact of climate on leachate quantity and quality is fairly well understood. In hot and humid climates, leachate production could be maximum compared to that generated in hot and arid climates. High levels of rainfall and porous soils create large quantities of leachates, although the concentrations of contaminants leached will be lower than in low rainfall areas. Evapotranspiration may also play a significant role in the overall water balance, particularly in hot and arid regions.

Although the hydrogeologic environments of any two landfills will have certain conceptual similarities, each landfill will exhibit factors unique to its setting which will greatly influence the nature and fate of leachates formed. Therefore, a three dimensional understanding of infiltration and groundwater movement is necessary for evaluating the suitability of candidate landfill sites or for planning control strategies at existing landfills.

Important factors include particle sizes and types of soils in the underlying strata, soil or other material used as a daily cover, and sizes and degree of compaction of wastes placed (Hughes, et al., 1971). Obviously, finely textured materials will allow for relatively low rates of leachate or gas movement, whereas, coarse materials or fractured bedrock will allow relatively easy passage of both liquids and gases.

These site specific uncertainties have been the primary contributors to deviations in leachate and gas characteristics from seemingly similar wastes. Moreover, they have often promoted an insistence on the application of restrictive management concepts, ranging from leachate removal and treatment to total landfill containment or waste encapsulation. Reliable and convincing hydrogeologic mapping will be increasingly required to offset growing concerns about short- and long-term impacts of landfills intended for the receipt of wastes from the industrial or municipal sectors. (This subject will be further addressed under the LEACHATE AND SOIL INTERACTIONS section of this report.)

INPUT WASTE CHARACTERISTICS

The majority of wastes disposed of in sanitary landfills are solid in nature, although the presence of municipal and industrial sludges is also common. Wastes originating from different source categories will contain different constituents which will also impart certain associated characteristics to the leachate produced.

As shown in Table 2, five major source categories can be identified; residential, agricultural, commercial, municipal, and industrial. Accordingly, residential and commercial wastes are comprised primarily of paper products (rubbish) ash, and food wastes. Agricultural wastes will include these products plus larger proportions of organic materials from crops and animals (agricultural wastes may also contain some potentially toxic material in the form of insecticides or pesticides). Industrial wastes will contain materials characteristic of the industry from which they originate.

TABLE 2. WASTE SOURCE CATEGORIES AND CORRESPONDING WASTE TYPES

Source Category	Major Waste Constituents
Residential	Rubbish, food and garden wastes, plastics, glass, ash
Agricultural	Crop and animal wastes, food wastes, rubbish, chemicals
Commercial	Rubbish, food wastes, construction/demolition debris, ash
Municipal	Rubbish, ash, food wastes, sewage sludge
Industrial	Biological and chemical sludges, rubbish, ash, construction/demolition debris

Several investigators have determined the relative composition of municipal solid wastes. As summarized in Table 3, the diversity of the results presented is indicative of the high potential for variance in the composition and relative proportions of wastes contributed from each source category. Nevertheless, from these analyses it can be expected that rubbish, food and garden wastes, and crop and animal residues will contribute organic compounds. Organic compounds will also be available from sewage sludges and certain industrial wastes. These wastes will also contribute to the moisture needed for leachate formation and biological activity leading to gas production. Ash wastes will contribute inorganic constituents, as will construction and demolition debris and the many types of industrial sludges and residues which constitute common sources of heavy metals.

TABLE 3. RANGE OF COMPOSITION OF MUNICIPAL SOLID WASTE*

Reference Source	(149)	(116)	(175)	(36)		(78)		(259)	
Component	Average	Average	Average	Range	Average	Range	Average	Range	Typical
Food Wastes	12	>25.1	25.0	8.8-12.8	10.7	4-9	7	6-26	15
Garden Wastes			0	5.8-17.0	10.4	1-10	5	0-20	12
Paper	39	44.5	50.0	>35.2-45.3	>40.6	45-57	50	25-45	40
Cardboard	7							3-15	4
Plastics		>22	>3.0	>4.2-5.2	>4.6			2-8	3
Rubber	2					4-9	6	0-2	1
Leather								0-2	1
Textiles	3	1.1	5.0	1.1-2.5	1.7	2-5	3	0-4	2
Plastic Film	2								
Wood	7		1.0	0.4-1.3	1.0	1-2	1	1-4	2
Glass	10	11.3	7.0	9.1-12.4	10.9	9-17	12	4-16	8
Metallics	8	8.7	4.0	8.0-8.6	9.0	6-15	10		
Tin Cans								2-8	6
Non-Ferrous Metals								0-1	1
Ferrous Metals								1-4	2
Dirt, Ashes, Brick, etc.	10	7.1	5.0	1.0-3.6	2.8	3-15	7	0-10	4
Moisture						21-35	27	15-40	20

*Percent by weight, wet weight basis

The codisposal of industrial sludges and residues with municipal, commercial, agricultural and/or residential wastes provides a potential source of toxic constituents. These constituents are usually inorganic (alkali and alkaline earth metals, heavy metals, nitrogen and sulfur compounds) but may also be organic in nature. Therefore, their migration into groundwater may pose health hazards and may actually inhibit or impede landfill stabilization or the performance of external leachate treatment processes. Nevertheless, due to the small generator exclusion in the Resource Conservation and Recovery Act (RCRA), small quantities of hazardous materials have been and are currently being codisposed in sanitary landfills and need to be considered as a recognized input. In these cases, *in situ* stabilization may occur at reduced rates as the leachate becomes more concentrated, and the extended stabilization period increases the opportunity for leachate migration from the landfill.

The degree of inhibition of the biologically mediated processes of stabilization within a landfill will depend upon the nature and quantities of potential inhibitors present. Recent research has demonstrated that appropriate combinations of industrial wastes with municipal refuse can reduce and/or eliminate the otherwise adverse effects of industrial wastes on stabilization (Bromley and Wilson, 1981; Jones and Malone, 1982; Chang, 1982). In some cases, codisposal of industrial wastes may contribute moisture or buffer capacity which encourages the onset of biological stabilization within the landfill (Kinman, *et al*., 1980; Kinman, 1982). Swartzbaugh, *et al*., (1978) reported that codisposal of several industrial wastes generally increased overall moisture content and caused a more rapid attainment of field capacity. However, experiments with petroleum wastes revealed a potential for the inhibition of leachate formation, and codisposal of battery wastes resulted in higher concentrations of leachate metals and other inorganic contaminants.

The differences in impact attributed to the industrial waste component during codisposal may also be a function of pH. Using small-scale leaching tests, Houle (1977) noted increased mobilization of metallic ions when leachate was used instead of distilled water. Similarly, Streng (1976) noted increased metal mobility during tests of codisposal of six selected industrial wastes. In contrast, Barber, *et al*., (1981) indicated that larger-scale studies revealed little evidence of increased metal leaching except at below pH 5. The authors speculated that this was due to attenuation by bicarbonate and sulfide precipitates and complexes. Similar observations have been recorded by Pohland, *et al*., (1981) and Walsh, *et al*., (1983), i.e., leaching of metals was initially attenuated by sulfide precipitation, followed by an increased mobility in the latter phases of biological stabilization due to possible complexation.

LANDFILL AGE (DEGREE OF STABILIZATION)

Landfill Stabilization Phases

The coupling of landfill age with leachate and gas production (quantity and quality) has been one of the seemingly most elusive challenges confronting designers, operators, and regulators of landfill disposal sites. The designer may conceive of operational features not responsive to requirements for leachate management as leachate is produced and changes in quality with time. Similarly, the treatment plant operator may be frustrated by the inability to adjust to these emerging circumstances, and the regulator may impose highly conservative and/or restrictive conditions in anticipation of these events, thereby stifling development and implementation of new and innovative technology.

In reality, most landfills receiving municipal solid waste proceed through a series of rather predictable events whose significance and longevity are largely determined by the previously mentioned climatological conditions, operational variables, management options, and control factors operative or being applied either external or internal to the landfill environment. Fortunately these events can be followed by certain leachate (and gas) analyses, selecting those parameters as major environmental factors that best describe certain conditions or "phases" of stabilization.

To direct the choice of analyses to be used to describe a particular phase of stabilization, it is necessary to recognize that a landfill exists throughout much of its active life as an anaerobic microbial process, analogous in concept to a batch digester, with limited inputs or outputs except for the refuse and moisture or eventual gas production and possible leachate migration, respectively. Using this analogy and recognizing that the functional retention time extends over a period of years rather than days, certain performance related and time dependent concepts emerge.

As with many anaerobic digestion systems, landfills experience an initial lag or adjustment phase which lasts until sufficient moisture has accumulated to encourage the development of a viable microbial community, the evidence of which is first observed in leachate quality when "field capacity" has been reached. Thereafter, further manifestations of waste conversion and stabilization may be reflected by changes in leachate and gas quality as stabilization proceeds through several more or less discrete and sequential phases, each varying in intensity and longevity according to prevailing operational circumstances. To illustrate this premise, the following five stabilization phases have been identified in terms of the principal events occurring during each (Pohland, et al., 1983).

Phase I: Initial Adjustment--

- Initial waste placement and preliminary moisture accumulates.
- Initial subsidence and closure of each landfill area.
- Changes in environmental parameters are first detected to reflect the onset of stabilization processes which are trending in a logical fashion.

Phase II: Transition--

- Field capacity is exceeded and leachate is formed.
- A transition from initial aerobic to anaerobic microbial stabilization occurs.
- The primary electron acceptor shifts from oxygen to nitrates and sulfates with the displacement of oxygen by carbon dioxide in the gas.
- A trend toward reducing conditions is established.
- Measurable intermediates such as the volatile organic fatty acids first appear in the leachate.

Phase III: Acid Formation--

- Intermediary volatile organic fatty acids become predominant with the continuing hydrolysis and fermentation of waste and leachate constituents.
- A precipitous decrease in pH occurs with a concomitant mobilization and possible complexation of metal species.
- Nutrients such as nitrogen and phosphorus are released and utilized in support of the growth of biomass commensurate with the prevailing substrate conversion rates.
- Hydrogen may be detected and affect the nature and type of intermediary products being formed.

Phase IV: Methane Fermentation--

- Intermediary products appearing during the acid formation phase are converted to methane and excess carbon dioxide.
- The pH returns from a buffer level controlled by the volatile organic fatty acids to one characteristic of the bicarbonate buffering system.
- Oxidation-reduction potentials are at their lowest values.
- Nutrients continue to be consumed.
- Complexation and precipitation of metal species proceed.
- Leachate organic strength is dramatically decreased in correspondence with increases in gas production.

Phase V: Final Maturation--

- Relative dormancy following active biological stabilization of the readily available organic constituents in the waste and leachate.
- Nutrients may become limiting.
- Measurable gas production all but ceases.
- Natural environmental conditions become reinstated.
- Oxygen and oxidized species may slowly reappear with a corresponding increase in oxidation-reduction potential.
- More microbially resistant organic materials may be slowly converted with the possible production of humic-like substances capable of complexing with and re-mobilizing heavy metals.

All of the major events selected to describe and separate these landfill stabilization phases are encountered at one time or another in landfills containing municipal refuse, provided that the associated microbially mediated processes have been augmented by a sufficiency of moisture and nutrients and are not being exposed to the inhibitory influences of toxic materials. Because the manifestations of these phases often overlap within the usual landfill setting, it has become customary to view them in a collective fashion. Unfortunately, this tends to obscure reality and limit understanding of the progression of events so requisite of design and operational attention. No landfill has a single "age", but rather a family of different ages associated with the various sections or cells within the landfill complex and their respective progress toward stabilization. Moreover, the rate of progress through these phases may vary depending on the physical, chemical and microbiological conditions developed within each section with time. For example, acid conditions established during acid formation may preclude the onset of active methane fermentation, microbial inhibition may be induced by the presence of toxic substances, or high compaction may restrict the movement of moisture and nutrients throughout the waste mass.

Indicator Parameters Descriptive of Stabilization Phases

There are certain indicator parameters or indices capable of being used to detect and describe the presence, intensity and longevity of each phase of landfill stabilization. Many of these apply to the analysis of leachate, so that their facility is most evident when leachate production has commenced. In addition, whether these analyses are physical, chemical or biological helps to determine their applications and interrelationships within an overall landfill perspective. For example, pH and ORP are physical-chemical parameters indicative of respective acid-base and oxidation-reduction conditions and critical to the proper evaluation of the acid formation and methane fermentation phases; COD and BOD_5 are chemical and biological parameters, respectively, but are both indicative of relative biodegradability; and, nitrogen and phosphorus are chemical parameters important to the determination of nutrient sufficiency and condition (aerobic/anaerobic) of a particular phase. Similar importance can be ascribed to other parameters which may reflect such factors as buffer capacity (alkalinity), potential inhibition (heavy metals), ionic strength/activity (conductivity), migration potential (chlorides), health hazards (bacteria and viruses) and oxidizing potential (nitrates and sulfates).

Ranges in intensity or concentration of these parameters will vary throughout the phases of stabilization, again depending upon the principal function of each phase as described and the physical influence of dilution with continuing ingress of moisture. This latter effect will tend to diminish concentrations during leachate analysis, but will not influence the total mass of leached constituents in time and space. Unfortunately, dilution effects are often poorly recorded, leading to analytical variances in magnitude and interpretation when analyses are based upon concentration alone. Nevertheless, there are data available in the literature which may be employed to provide general ranges of intensity and concentration of these indicator parameters throughout those landfill stabilization phases when leachate is available for analysis. Table 4 provides such a compilation for the four previously defined landfill phases during which leachate and gas analyses are critical for characterization and interpretation. These data have been derived and arranged from literature accounts of a diverse group of primarily laboratory or pilot-scale landfill simulations reviewed herein and presented to indicate the magnitude of ranges encountered. Scrutiny of these data indicates some obvious overlap between phases and also some contradictions of the relatively discrete descriptions presented previously for each landfill stabilization phase.

To better demonstrate this ability to match changes in leachate (and gas) analyses with stabilization phases, and to use the results of such a procedure to provide both didactic and operational interpretations of landfill behavior, data from previously reported pilot-scale investigations of accelerated landfill/leachate stabilization with leachate recycle (Pohland, 1980) have

TABLE 4. LANDFILL LEACHATE AND GAS CONSTITUENT CONCENTRATION RANGES ENCOUNTERED IN THE LITERATURE* AND THEIR RELATIVE SIGNIFICANCE TO THE DEGREE OF LANDFILL STABILIZATION

Leachate or Gas Constituent	Phase of Biological Stabilization				Overall Range (All Phases)
	Transition Phase	Acid Formation Phase	Methane Fermentation Phase	Final Maturation Phase	
Biochemical Oxygen Demand (BOD_5)	100-10,900 Influence of dilution and aerobic solubilization of waste organics	1,000-57,700 Accumulation of biodegradable organic acids due to methanogenic lag	600-3,400 Conversion of biodegradable organics to gaseous end products (CH_4 and CO_2)	4-120 Influence of high-molecular weight organic residuals (humics, fulvics)	4-57,700
Chemical Oxygen Demand (COD)	480-18,000 Trending in a similar fashion to BOD_5	1,500-71,100 Trending in a similar fashion to BOD_5	580-9,760 Trending in a similar fashion to BOD_5	31-900 Higher influence of residual organics than in BOD_5 assay.	31-71,700
Total Organic Carbon (TOC), mg/l	100-3,000 Beginning to appear as a result of aerobic solubilization	500-27,700 Increasing rapidly; accumulation due to methanogenic lag	300-2,230 Conversion of volatile acids to methane; decrease in aqueous carbon	70-260 Influence of higher molecular weight organics (humics, fulvics)	70-27,700
Total Volatile Acids (TVA), mg/l as Acetic Acid	100-3,000 Or just beginning to appear as a result of solubilization	3,000-18,800 Solubilization of organic polymers to monomers; beta-oxidation to volatile acids	250-4,000 Conversion of fatty acids to fatty acid; fermentation of acetic acid to methane	Essentially absent - methanogenic system undersaturated	0-18,800
BOD_5/COD Ratio	0.23-0.87 Increasing biodegradability of organics due to solubilization	0.4-0.8 High biodegradability	0.17-0.64 Decreasing biodegradability due to methanation	0.02-0.13 Low degree of biodegradability	0.02-0.87
COD/TOC Ratio	4.3-4.8 Low oxidation state or organics	2.1-3.4 Low to moderate oxidation state of organics	2.0-3.0 Moderate to high oxidation of organics	0.4-2.0	0.4-4.8
Total Kjeldahl Nitrogen (TKN) mg/l	180-860	14-1,970 May be low due to microbial assimilation of nitrogenous compounds	25-82 Low due to microbial assimilation of nitrogenous compounds	7-490	7-1,970
Nitrate Nitrogen (NO_3^--N), mg/l	0.1-5.1 Increasing due to oxidation of ammonia	0.05-19 Decreasing due to reduction to ammonia or N_2 gas	Absent Complete conversion to ammonia or N_2 gas	0.5-0.6	0-51
Ammonia Nitrogen (NH_3-N) mg/l	120-125	2-1,030 Increasing due to nitrate reduction and protein breakdown	6-430 Decreasing due to biological assimilation	6-430	2-1,030
NH_3/TKN Ratio	0.1-0.9	0-0.98 Protein breakdown; biological assimilation	0.1-0.84	0.5-0.97	0.1
Total Phosphate ($PO_4^=$-P), mg/l	0.6-1.7	0.2-120 Biological assimilation and metal complexation	0.7-14 Low due to biological assimilation	0.2-14	0.2-120

TABLE 4 (Continued)

Total Alkalinity, mg/l as $CaCO_3$	200-2,500	140-9,650 Increasing due to volatile acid formation and bicarbonate dissolution	760-5,050 Decreasing due to volatile acid removal	200-3,[illegible]20	140-9,650
Solids (TS), mg/l	2,450-2,050	4,120-55,300 Increasing due to solubilization or organics and mobilization of metals	2,090-6,410	1,460-4,640	1,460-55,300
pH	6.7	4.7-7.7 Low due to volatile acid accumulation	6.3-8.8 Increasing due to volatile acid removal and bicarbonate dissolution	7.1-8.8	4.7-8.8
Oxidation-Reduction Potential (ORP), mV	+40 to +80	+80 to -240 Decreasing due to the depletion of oxygen	-70 to -240	+97 to +163	-240 to +16
Copper, mg/l	0.085-0.39	0.005-2.2	0.03-0.18 Decreasing (complexation)	0.02-0.56	0.005-2.2
Iron, mg/l	68-312	90-2,200	115-336 Decreasing (complexation)	4-20	4-2,200
Lead, mg/l	0.001-0.004	0.01-1.44	0.01-0.1 Decreasing (complexation)	0.01-0.1	0.001-1.44
Magnesium, mg/l	66-96	3-1,140	81-505 Decreasing (complexation)	81-190	3-1,140
Manganese mg/l	0.6	0.6-41	0.6 Decreasing (complexation)	0.6	0.6-41
Nickel, mg/l	0.02-1.55	0.03-79	0.01-1.0 Decreasing (complexation)	0.07	0.02-79
Potassium, mg/l	35-2,300	35-2,300	35-2,300	35,2,300	35-2,300
Sodium, mg/l	20,7,600				20-7,600
Zinc, mg/l	0.06-21	0.65-220	0.4-6.0	0.4	0.06-220
Total Coliform, CFU/100 ml	10^0 to 10^5	10^0 to 10^5	Essentially absent	Absent	$0-10^5$
Fecal Coliform, CFU/100 ml	10^0 to 10^5	10^0 to 10^5	Essentially absent	Absent	$0-10^5$
Fecal Streptococci, CFU/100 ml	10^0 to 10^6	10^0 to 10^6	Essentially absent	Absent	$0-10^6$

TABLE 4 (Continued)

Viruses, PFU/100 ml	--	**Essentially absent**	**Essentially absent**	**Essentially absent**	**Absent**
Conductivity, μmhos/cm	2,450-3,310	1,600-17,100 Increasing due to mobilization of metals	2,900-7,700 Decreasing due to metals complexation with sulfides	1,400-4,500	1,400-17,100
Chloride (Cl^-), mg/l	30-5,000 Biologically stable; good indicator of washout	30-5,000 Stable; good hydraulic tracer	30-5,000 Stable; good hydraulic tracer	30-5,000 Stable; good hydraulic tracer	30-5,000
Sulfate $(SO_4^=)$, mg/l	10-458 Increasing due to aerobic oxidation	10-3,240 Increasing initially due to aerobic solubilization then decreasing as anaerobic conditions are estabilished	Absent Complete conversion to sulfides	5-40 Reappearing due to aerobic oxidation	
Sulfide $(S^=)$, mg/l	Essentially absent	0-818 Beginning to appear and increasing due to sulfate reduction under anaerobic conditions	0.9 Low due to heavy metal precipitation	Absent	
Cadmium, mg/l	190-490	70-3,900	76-490 Decreasing due to complexation and precipitation	76-254	70-3,900
Chromium, mg/l	0.023-0.28	0.06-18	0.05 Decreasing due to complexation, precipitation with sulfides	0.05	0.02-18
Methane, %	Essentially absent (aerobic metabolism)	Very low (<1%); Transition to anaerobic metabolism	30-60 Suitable for energy recovery	0-<10 Decreasing due to substrate limitation and reversion to aerobic metabolism	0-60
Carbon Dioxide, %	0-10 Product of aerobic decomposition of organics	10-30 Increasing due to waste decomposition	30-60 Decreasing to <5% as methanogenesis increases	<40 Aerobic metabolism	0-60
Nitrogen Gas, %	70-80 Influence of trapped air	60-80 Decreasing due to dilution with CO_2	<20 Artefact of trapped air; denitrification	>20 Increasing due to introduction of air	<20-80
Oxygen, %	20 Influence of trapped air	0-5 Decreasing due to aerobic utilization; shift towards anaerobic metabolism	0-5 Disappearing as methanogenesis increases	>5 Increasing due to introduction of air	0-20
Hydrogen, %	Essentially absent in the presence of oxygen	0-2 Beginning to appear as oxygen is depleted; accumulates until methanogenesis occurs	<0.1 Maintained at low levels by methanogenesis; difficult to measure	0-2 Essentially absent	

[*] Ranges of constituent concentrations were collected from the references and data presented in the Appendices.

been reproduced and presented in Figure 3 for COD, total volatile acids (TVA), pH, gas production and composition, and ORP; parameters considered as major environmental factors within the landfill environment. Since these data were obtained during municipal refuse stabilization after leachate had been produced for recycle, they cover a time period extending from transition (Phase II) to final maturation (Phase V), with the manifestations of acid formation (Phase III) and methane fermentation (Phase IV) being most pronounced.

In reality, most detectable landfill stabilization is accountable to the processes occurring during Phases III and IV. With leachate recycle, the consequences are magnified and reflected in the indicator parameters over a more contracted time interval than normally encountered at conventionally managed landfills. Accordingly, high concentrations of organic contaminants, represented by COD (shaded area) and TVA analyses, appeared inthe leachate soon after leachate recycle was commenced (Time 0) as indicated in Figure 3. Thereafter, the magnitude of these same parameters decreased as gas production increased during methane fermentation, changing the initial ambient gas composition to one dominated by methane and carbon dioxide. Similarly, the formation and subsequent microbial conversion of volatile acids caused an initial increase and decrease in COD. All of these changes are similar to those occurring in many anaerobic biological treatment systems as they progress sequentially through acid and methane fermentation phases. In addition, since the experimental landfill used in these studies was constructed similar to a discrete cell at a conventional landfill site, the progress of stabilization (although accelerated by refuse shredding and leachate recycle) reflected the landfill aging process for an analogous section of a landfill where the two most active phases (Phases III and IV) were essentially completed in about one year.

In actual landfills, the time periods associated with each phase and the quality and quantity of leachate and gas will vary according to landfilling procedures, the nature of the wastes, the amount of moisture allowed as input to the landfill and closure and post-closure methods eventually applied. Therefore, the time scale and concentration intensity for each of the five phases indicated will vary from site to site. Nevertheless, Figure 3 serves to illustrate the trends to be expected in the quality of both leachates and gases produced with time. A careful analysis of associated project data from a particular site can give a good indication of the existing "phase". Moreover, a historical data base may allow prediction of lengths of phases and facilitate a better planning and management of both leachate and gas handling technologies as well as long-term maintenance.

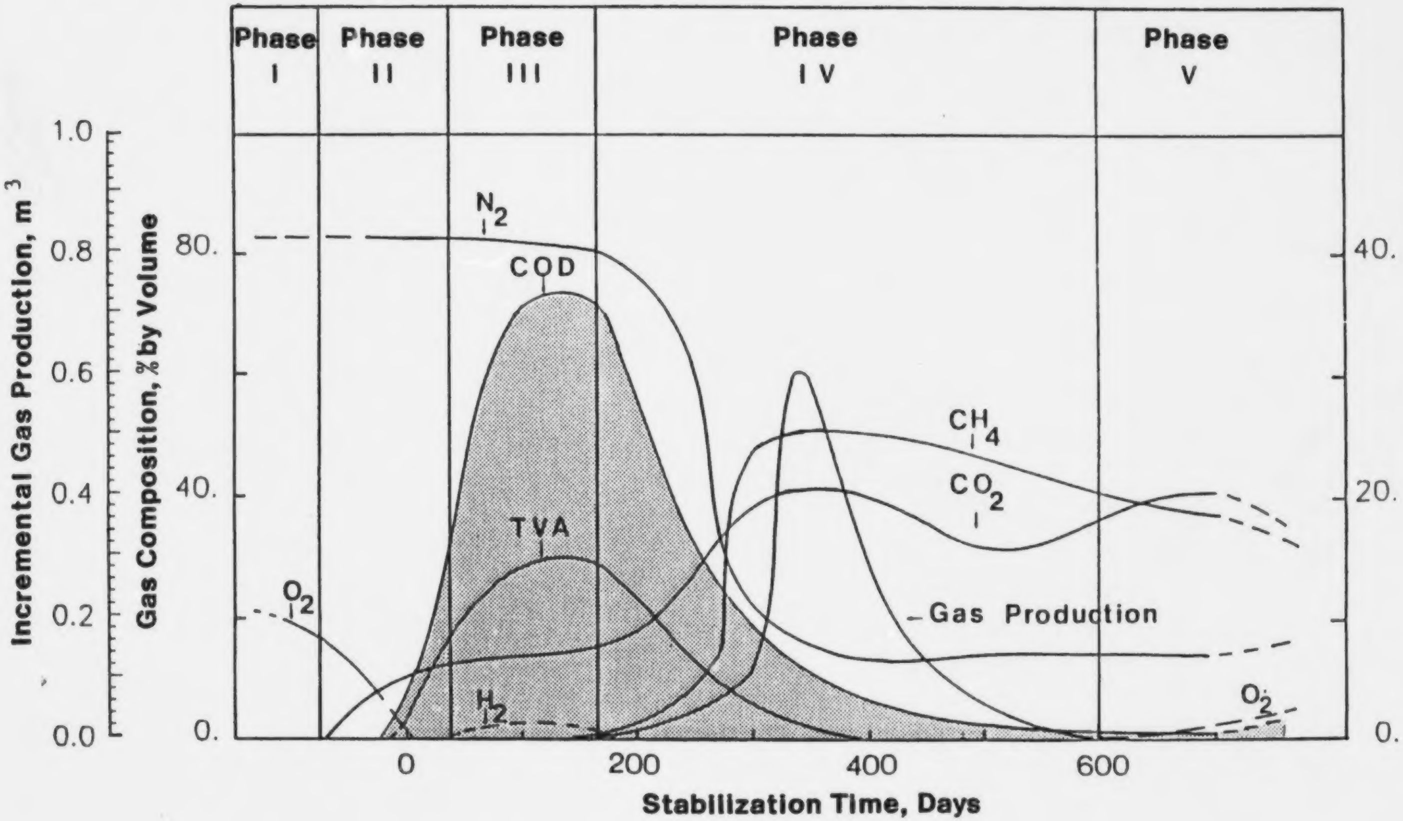

Figure 3. Changes in Selected Indicator Parameters During the Phases of Landfill Stabilization (after Pohland, 1980 and Pohland, et al., 1983b).

SECTION 6

TREATMENT OF LEACHATES FROM SANITARY LANDFILLS

GENERAL PERSPECTIVE

Most processes commonly employed for treatment of industrial wastewaters have been tested for treatment of landfill leachates. These include the traditional aerobic and anaerobic biological processes as well as a variety of physical-chemical processes. Some of these processes are intended primarily for the removal of organic contaminants, while others are best suited for inorganics removals. Moreover, process performance in each case is related to the chemical nature of the leachate utilized as influenced by the age of the landfill as previously described and the miscellaneous factors previously described. Accordingly, certain of the processes may be also used in pre- or post-treatment applications.

The purpose of this section is to present a coordinated review of biological and physical-chemical treatment process capabilities, following an approach which segregates processes into bench-, pilot- and full-scale categories and leachates into low-, medium-, and high-strength categories. The review was organized to present the biological processes first (aerobic, then anaerobic), followed by physical-chemical processes including applications of coagulation, oxidation, ionizing radiation, ion-exchange, adsorption, and reverse osmosis.

AEROBIC BIOLOGICAL TREATMENT OF LANDFILL LEACHATE

The operation and evaluation of biological treatment processes are dependent upon the ability to monitor and control certain process variables. Considering carbon as the limiting nutrient in biological treatment systems, the design and operational variables of primary interest are those which reflect the rates of carbon utilization exhibited by a given cellular mass or reactor volume. The corresponding rates of biomass generation are also of interest with respect to the maintenance of a stable biological population and sludge disposal considerations.

Four kinetic parameters are generally used to describe the growth of microorganisms in response to the availability of a limiting substrate. These include the maximum specific cell growth rate (μ_{max}), the cell decay coefficient (b), the saturation coefficient (K_s), and the cell yield (Y). Operation in continuous culture requires a dynamic balance of substrate and cellular variations. In addition, substitution of classical Monod kinetic expressions containing the four variables mentioned above into a mass balance expression gives rise to several operational parameters. These include the mean cell residence time (θ_c), the volumetric organic loading rate (OLR), and the food to microorganism ratio (F/M) which is a cellular organic loading rate.

Process performance in response to manipulation of these operating variables is evaluated by comparing effluent organic concentrations to influent concentrations and to existing effluent limitations. Therefore, the treatment evaluations presented reflect COD and BOD_5 removals as well as effluent COD and BOD_5 concentrations. In addition, nitrification/denitrification, metals removal, and sludge characteristics, also important considerations in the treatment and disposal of leachates, are discussed in each treatment section depending upon information available.

Bench-Scale Aerobic Treatment Studies

A list of references pertaining to aerobic biological treatment of landfill or lysimeter leachates is presented in Table 5. The reactor configurations, research objectives and operating protocols associated with each study are also presented in the table.

Activated Sludge--

Activated sludge and its process variations have become well established for the treatment of municipal and many industrial wastewaters. Its wide ranging success in treating these wastewaters has encouraged a number of preliminary evaluations of its effectiveness in treating leachate. However, due to the wide variation in quality of leachates and in activated sludge operational protocols, results from these studies tended to be somewhat diverse. Therefore, comparisons of process performance in terms of effluent organic concentrations and percentage removals were arranged to reflect results from three different influent organic strength categories. Accordingly, the effects of θ_c, organic loading rates, and other process variables were determined for low-, medium-, and high-strength leachate categories and concomitant concentration ranges indicated in Table 6.

TABLE 6. LEACHATE ORGANIC STRENGTH CATEGORIES

Leachate Strength Category	Concentration Ranges, mg/l COD Basis	 BOD_5 Basis
Low-Strength	<1,000	220-750
Medium-Strength	1,000-10,000	750-1,500
High-Strength	10,000	1,500-36,000

Effect of Mean Cell Residence Time (θ_c)--Following the segregation of bench-scale activated sludge treatment data into the categories listed in Table 6, the influence of mean cell residence time on process performance was investigated by plotting θ_c versus effluent BOD_5 and COD concentrations and versus percentage removals. The resulting performance of bench-scale activated sludge units at 22-25°C is illustrated in terms of BOD_5 in Figure 4. The data presented in Figure 4 (summarized in Appendix Table A-1) suggest that the limiting θ_c, defined as that incurring an organic removal efficiency of

TABLE 5. BENCH-SCALE RESEARCH ON AEROBIC LEACHATE TREATMENT PROCESSES

REFERENCE	PROCESS*	PROCESS DESCRIPTION	RESEARCH OBJECTIVE(S)	LEACHATE SOURCE**
19,20, 195	AS	Complete-mix, continuous flow, extended aeration reactor system, daily fill and draw reactor operation.	Effect of θ_c and BOD_5 and COD loading on BOD and COD removal efficiencies.	Landfill
28	AS	Complete-mix, continuous flow reactor operation.	Effect of phosphorus addition, influent dilution and θ_c on BOD_5 and COD removal efficiencies; determine kinetic parameters.	Landfill and lysimeter
35,143	AS AL RBC TF	AS: Complete-mix, batch and continuous flow reactor; AL: Complete-mix, continuous flow reactor operation; RBC: Plug flow, continuous flow reactor operation; TF: Complete-mix, continuous flow reactor operation using plastic contact media.	Effect of COD loading and influent COD concentration on COD removal efficiency; influence of chemically pretreated influent.	Landfill
26	AS AL	Not given.	Effect of θ_c on BOD_5 efficiencies and metal removal.	Landfill
40,42,44, 45,70	AL	Complete-mix, daily fill and draw extended aeration reactor system.	Effect of τ on COD and TOC removal; nutrient addition effect; metals removal; sludge characteristics.	Landfill and lysimeter
53,54,97	AS	Complete-mix, continuous feed, reactor operation	Effect of θ_c on BOD_5, COD, and TOC removal efficiency, kinetic parameter determination.	Landfill
118,119	AS	Complete-mix, daily fill and draw reactor operation.	Effect of temperature and θ_c on organic removal efficiency.	Lysimeter
151	AS	Complete-mix, continuous flow reactor operation.	Effect of θ_c on COD and TOC removal efficiencies; metal removal efficiency; determine kinetic parameters.	Landfill and lysimeter
193,194	AS	Complete-mix, continuous flow reactor operation.	Determine kinetic parameters.	Lysimeter
205,206,207	AS	Complete-mix, continuous flow reactor operation.	Effect of θ_c on BOD_5, COD, TOC and metal removal efficiencies; determine kinetic parameters.	Landfill and lysimeter

TABLE 5 (Continued)

230	AS	Not given.	Effect of Θ_c and BOD_5 and COD loading on BOD_5 and COD removal efficiencies; effect of nutrient adjustment.	Landfill
28,232	AS	Complete-mix, continuous flow reactor operation.	Iron removal using ferrous iron metabolizing bacteria.	Lysimeter
230	AS	Complete-mix, continuous flow reactor operation.	Effect of chemical pretreatment on COD removal efficiency.	Lysimeter
244	AS AL	Complete-mix, continuous flow reactor operation.	Effect of temperature, Θ_c, BOD_5 loading, and BOD/COD on BOD_5 and COD removal efficiency; compare results of full-scale and bench-scale studies.	Landfill
260,261	AS	Complete-mix, daily fill and draw reactor operation.	Determine nutrient requirements for BOD_5 and metal removal efficiencies.	Lysimeter
269,270,271, 176	AS	Complete-mix, daily fill and draw, extended aeration reactor system.	Effect of Θ_c and F/M on BOD_5, COD and metal removal efficiencies; determine kinetic parameters.	Lysimeter
272	AL	Complete-mix, continuous flow reactor operation.	Effect of Θ_c on BOD and TOC remvoal efficiencies.	Landfill
288,289, 290	AS	Complete-mix, daily fill and draw reactor operation.	Effect of temperature and Θ_c on BOD_5 and COD removal efficiencies; effect of Θ_c on effluent polishing.	Lysimeter
291	AS	Complete-mix, fill and draw reactor operation.	Effect of fill and draw cycle on BOD_5 and COD removal efficiencies and sludge characteristics.	Lysimeter
168	AL	Complete-mix, fill and draw reactor operations.	Effects of Θ_c on organics, metals and nitrogen conversion.	Landfill

*AS = Activated Sludge
AL = Aerated Lagoon
TF = Trickling Filter
RBC = Rotating Biological Contactor

**All leachate sources involve the use of municipal solid waste

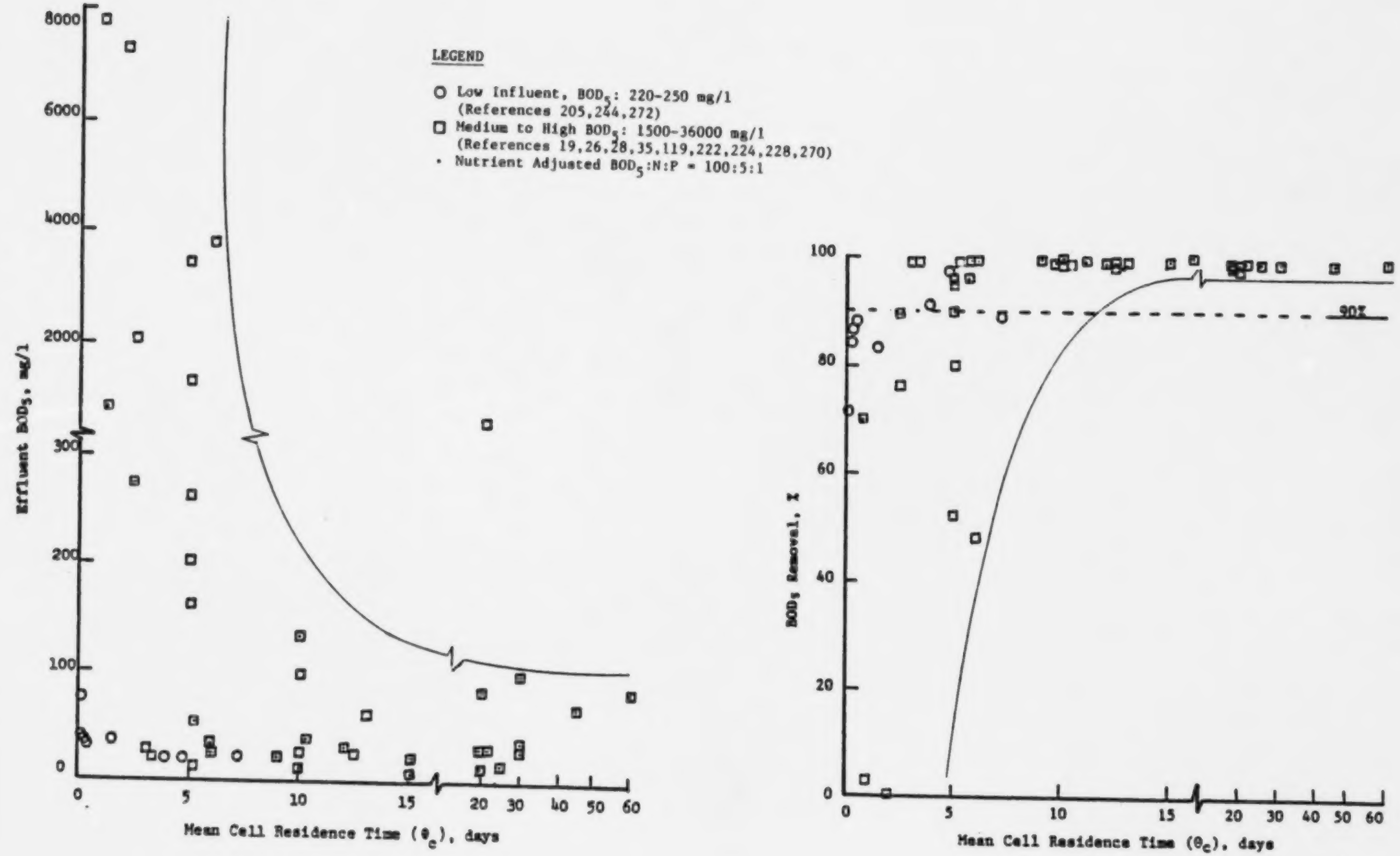

Figure 4. Relationship Between Θ_c and BOD_5 Removal for Bench-Scale Activated Sludge Studies

90% or greater and/or an effluent BOD_5 of 200 mg/l or less, ranged between 6 to 10 days. (It should be noted that many researchers did not include the BOD_5 assay in their monitoring programs. The COD and/or TOC analyses were generally preferred and, in spite of the advantages in simplicity and accuracy offered by these analyses, they did not necessarily reflect differences in leachate biodegradability.)

The effects of θ_c on process performance is illustrated on a COD basis in Figure 5. The data presented, also summarized in Appendix Table A-1, similarly suggest a limiting θ_c of 6 to 10 days. Compared with BOD_5-based analysis, the COD data suggest that significant process improvement may be available using extended retention times (10-12 days). However, even at retention times exceeding 10 days, effluent COD concentrations typically remained above 300 mg/l and effluent BOD_5 concentrations ranged from 10 to 100 mg/l. Residual COD may be attributed to refractory organics such as humic- and fulvic-like substances (Chian and DeWalle, 1977a, 1977b; Chang, 1982).

In addition to segregating the data based on influent concentration, the data were also divided into three biodegradability ranges with BOD_5/COD ratios of <0.50, 0.50-0.75, and >0.75 being characteristic of low-, medium-, and high-strength leachates, respectively. The data were similarly divided with COD/TOC <2.0 for low-strength, 2.0 < COD/TOC < 3.0 for medium-strength, and COD/TOC >3.0 for high-strength leachate. Percent COD removal was used to evaluate the effects of biodegradability on treatment performance, since it was the most commonly analyzed indicator of organic content in the leachates.

As anticipated, examination of the effects of θ_c and biodegradability on process performance using these ratios (Figure 6) indicates that the higher-strength leachates were more amenable to treatment at lower retention times than lower-strength leachates. As before, a limiting θ_c in the range of 5 to 10 days was suggested by the COD removal data.

Organic Loading Effects--The organic loading applied to bench-scale activated sludge processes was the second operational variable evaluated with respect to its effects upon effluent organic content (BOD_5, COD) and removals. As in the θ_c evaluation, the performance data were segregated into low-, medium-, and high-strength leachate categories using both influent BOD_5 and COD concentrations and BOD_5/COD and COD/TOC ratios.

The influences of two kinds of organic loading rate on activated sludge process performance were evaluated. The first was the volumetric organic loading rate, which is based upon the hydraulic retention time and is commonly referred to as the organic loading rate (OLR). The second loading rate is based upon the mass of microorganisms in the reactor as well as the hydraulic retention time. This latter loading rate is commonly referred to as the food-to-mass (microorganism) ratio, F/M.

Illustrations of the influence of OLR on BOD_5 and COD are presented in Figures 7 and 8, respectively. Data presented in these figures (Appendix Table A-2) do not exhibit very clear trends, although it may be suggested that the limiting OLR was on the order of 1 to 2 kg $BOD_5/m^3\cdot$day. The COD data are particularly diverse, making an analysis of process trends on this basis difficult. Even when further segregated into biodegradability categories using BOD_5/COD and COD/TOC ratios (Figure 9), no clear trend is discernible

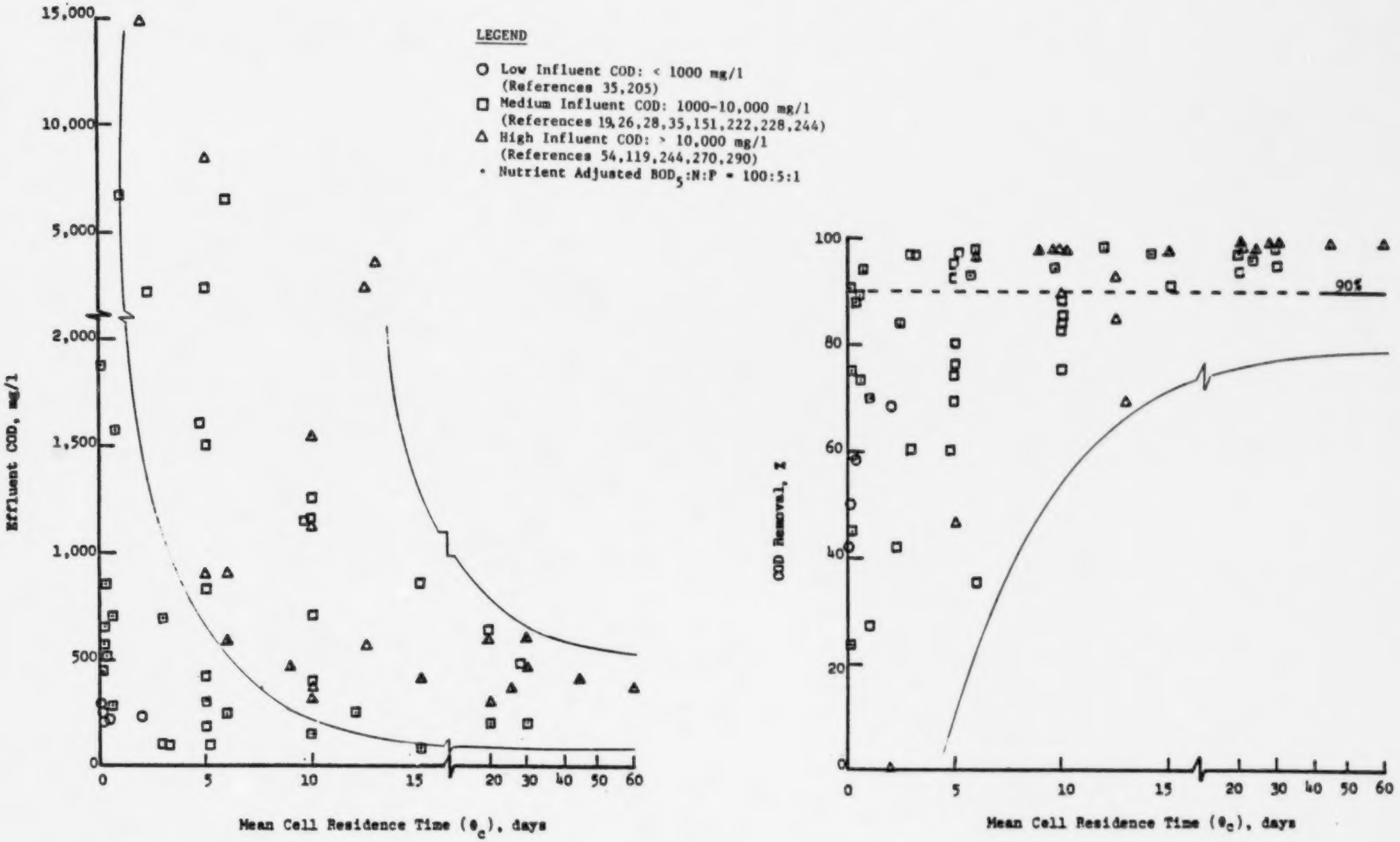

Figure 5. Relationship Between θ_c and COD for Bench-Scale Activated Sludge Studies

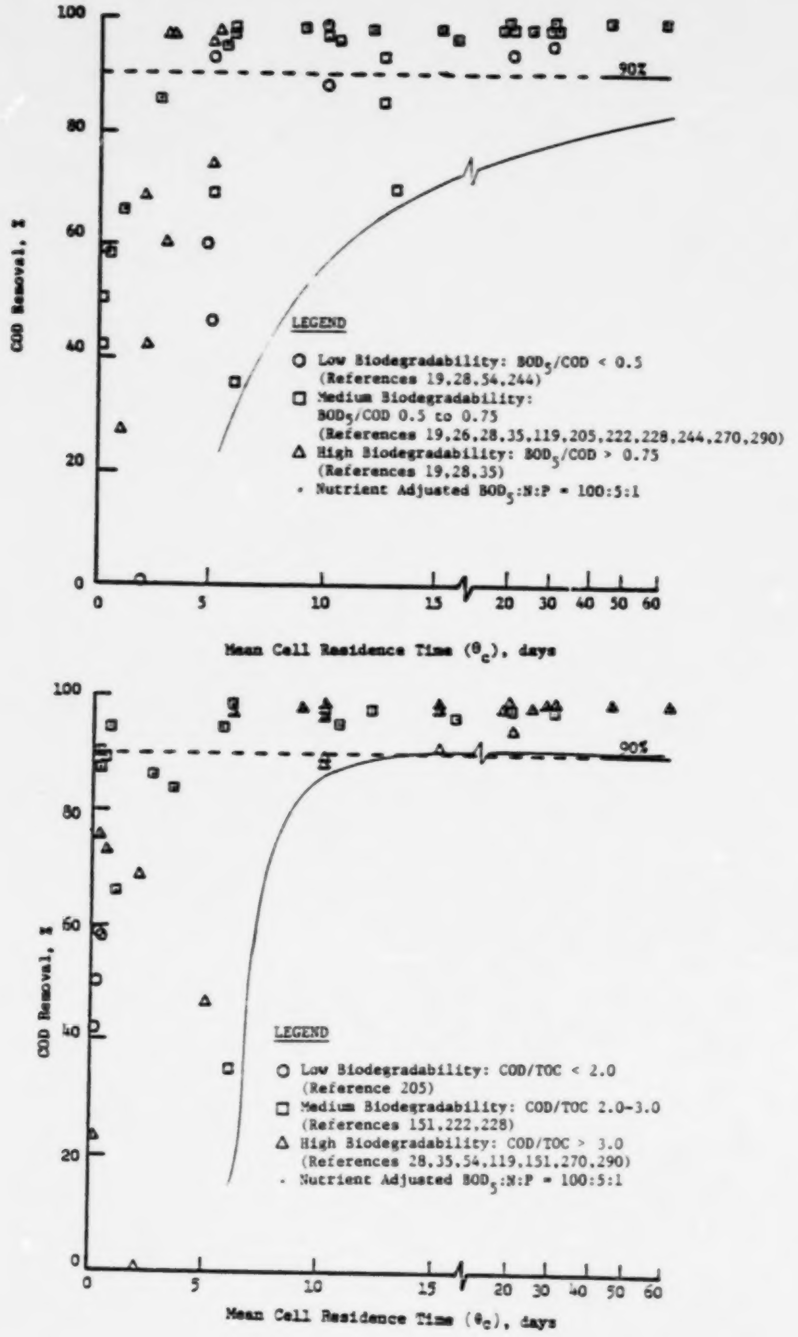

Figure 6. Comparison of θ_c vs. COD Removal Data Segregated According to Biodegradability Ratios BOD/COD and COD/TOC

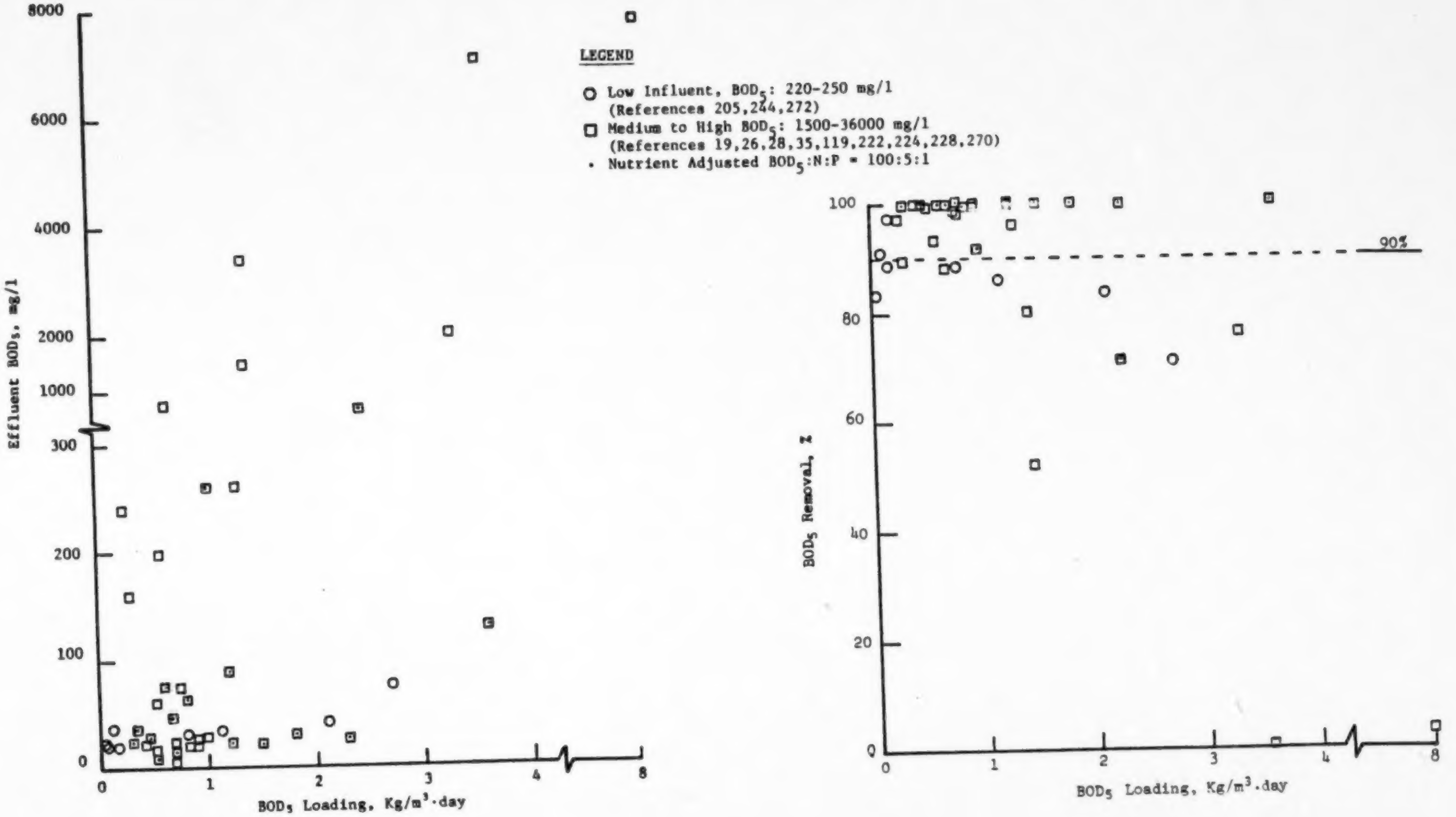

Figure 7. Relationship Between Organic Loading Rate and BOD5 Removal for Bench-Scale Activated Sludge Studies

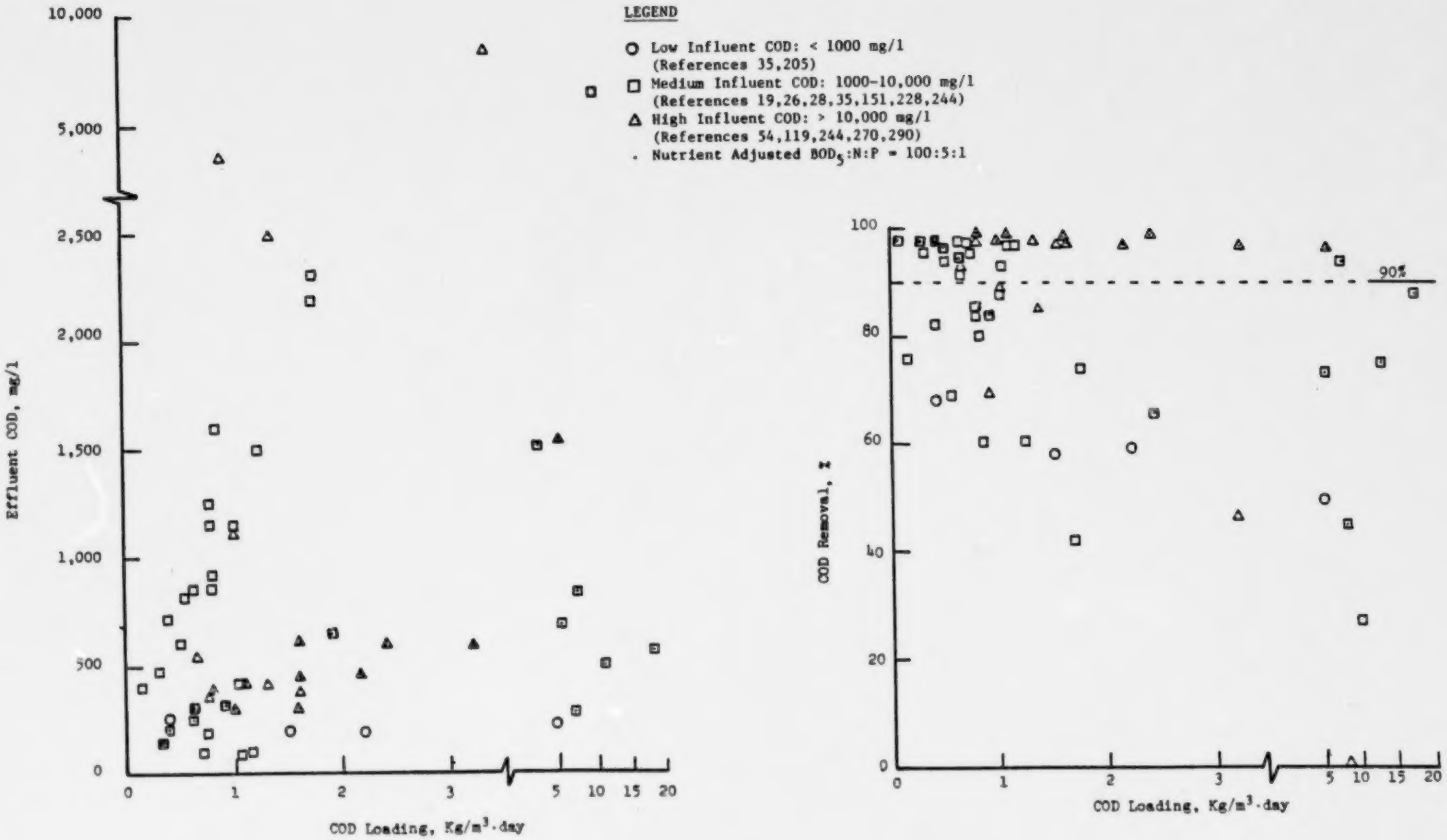

Figure 8. Relationship Between Organic Loading Rate and COD Removal for Bench-Scale Activated Sludge Studies

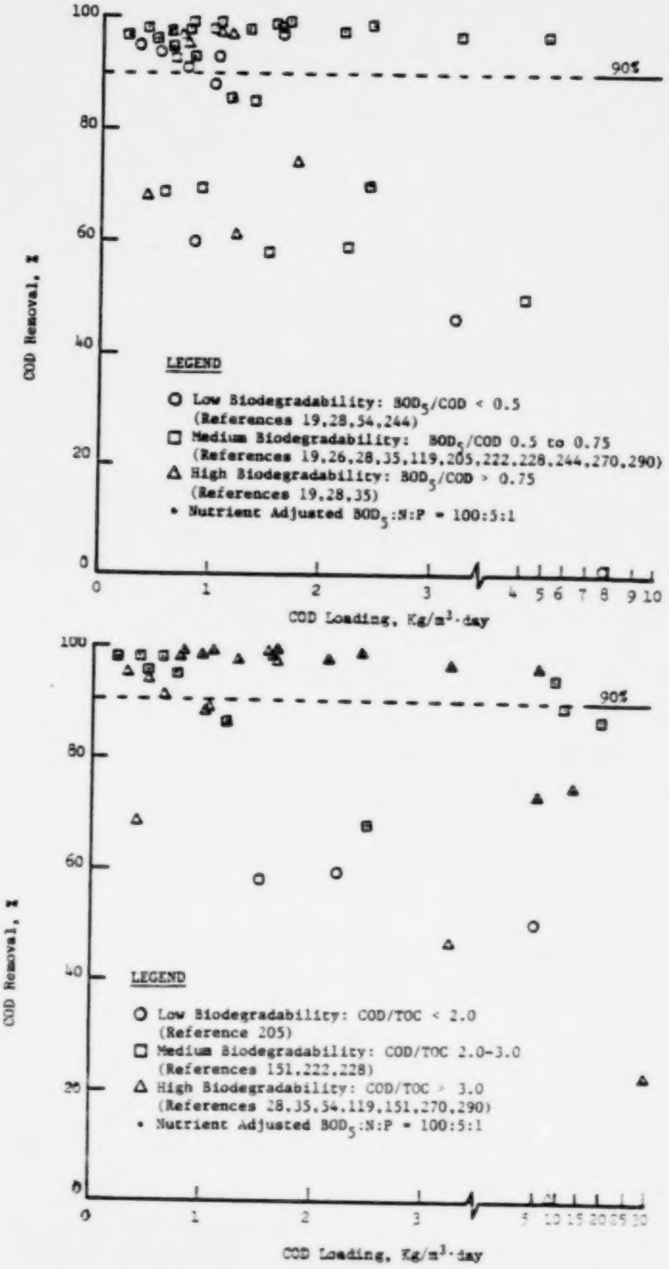

Figure 9. Relationship of Organic Loading Rate vs. COD Removal for Data Segregated by Biodegradability Ratios

with the COD loading data. However, successful operation has been demonstrated for both medium- and high-strength nutrient-amended leachates at loading rates up to 15 kg COD/m^3·day. For raw leachates, successful operation was only evident for loading rates approaching 2 kg COD/m^3·day.

The effects of varying F/M are represented in Figures 10 and 11 for BOD_5 and COD loadings, respectively. Limited F/M data (Appendix Table A-3) were available since not all researchers included mixed liquor volatile suspended solids (MLVSS) analyses in their monitoring programs. The BOD_5 data which are available (Figure 9) suggest that the limiting F/M may be on the order of 0.2 to 0.4 kg BOD_5/kg MLVSS·day. However, successful operation using nutrient-adjusted leachate has been demonstrated at loading rates beyond 0.5 kg BOD_5/kg MLVSS·day. Examination of the COD data (Figure 11) suggests similar trends; effluent COD values increased to above 700 mg/l at loading rates in excess of 0.4 kg COD/kg MLVSS·day. COD removals for medium- and high-strength leachates remained above 90% at this loading rate. As demonstrated using the OLR, nutrient amendments allowed for successful operation up to 1 kg COD/kg MLVSS·day.

Effects of Temperature--The effects of temperature on the performance of bench-scale activated sludge units were evaluated by comparing effluent COD concentrations and percentage removals at temperatures ranging from 5° to 25°C. However, since these performance parameters are also dependent on θ_c and the influent concentration, the data shown in Figure 12 are also compared with regard to these variables. Accordingly, the data are grouped into the three influent strength categories previously used and into three θ_c categories as well. As indicated in Figure 12, these include: $\theta_c < 4$ days; $\theta_c = 6$ to 10 days, and, $\theta_c > 12$ days.

Due to these and other operational variables, the effects of temperature are not clearly discernible, although the trend appears to be an expected increase in organic removal with increasing temperature. Successful operation has been demonstrated for $\theta_c > 6$ days at temperatures as low as 5°C. However, lower effluent COD concentrations resulted at the higher temperatures.

Heavy Metal and Alkaline Earth Metal Removal--Some researchers have included metals analyses in their monitoring protocol to evaluate the effectiveness of activated sludge in removing these constituents from leachates. As shown in Table 7 and Appendix Table A-4, the activated sludge process was effective in removing the majority of the heavy metals monitored. In particular, zinc, iron, manganese and cadmium were removed by 95% or greater. Chromium and lead were also fairly well removed (80-90%). However, nickel removals were generally low and on the order of 60%. Metal removal during aerobic treatment may be enhanced by the oxidation of metals, e.g., Fe^{+2} to Fe^{+3}, to forms which precipitate more easily at the pH of ranges of 8 to 9 commonly encountered during activated sludge leachate treatment.

The alkaline earth metals were removed, but to a lesser degree than observed for the heavy metals during normal activated sludge operation. As again shown in Table 7 and Appendix Table A-4, calcium and magnesium removals ranged from 3 to 99%, but were typically in the range of 40 to 70%. Potassium and sodium removals were typically on the order of 20 to 40%.

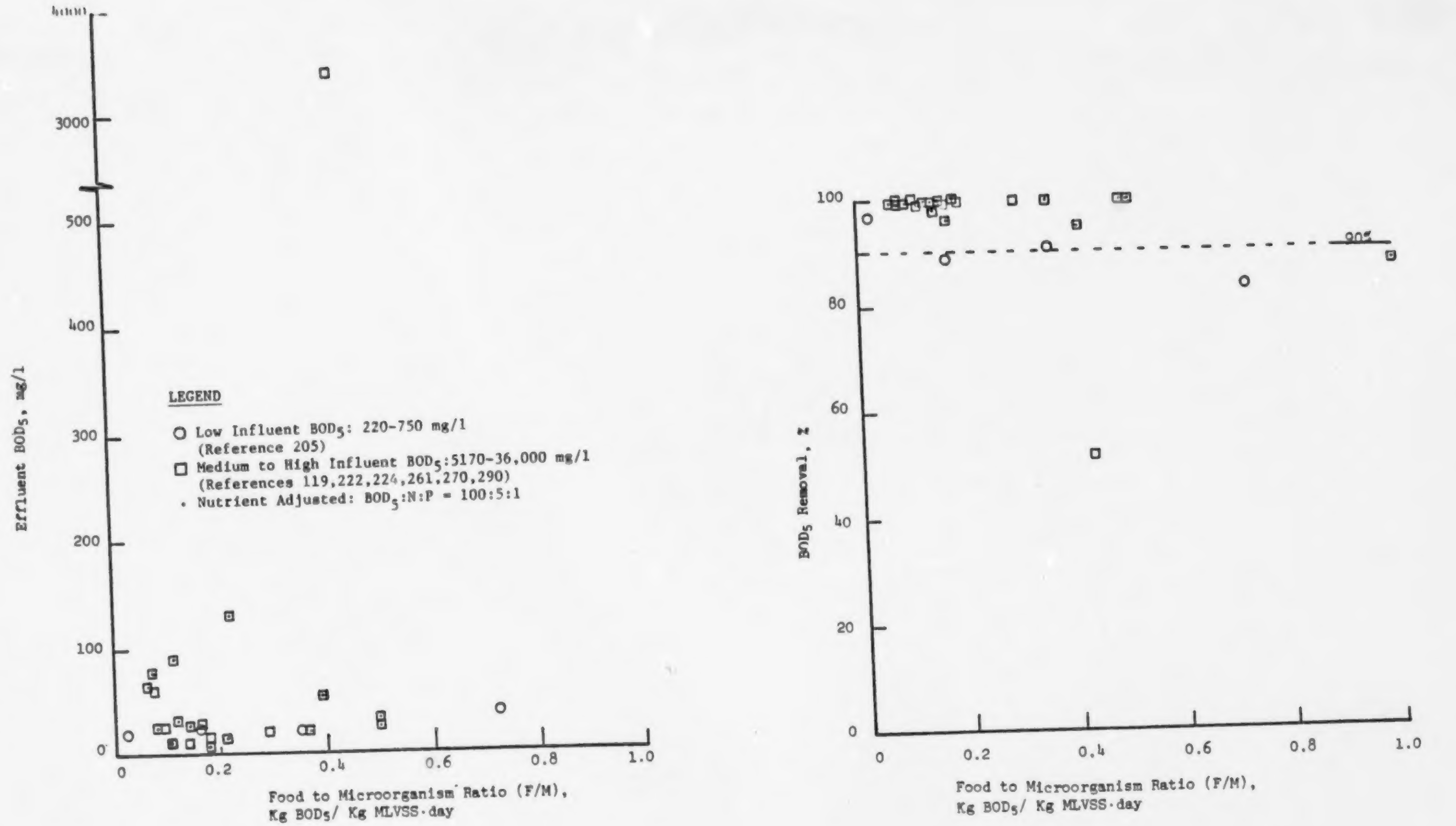

Figure 10. Relationship Between Food to Microorganism Ratios (F/M) and BOD_5 Removal for Bench-Scale Activated Sludge Studies

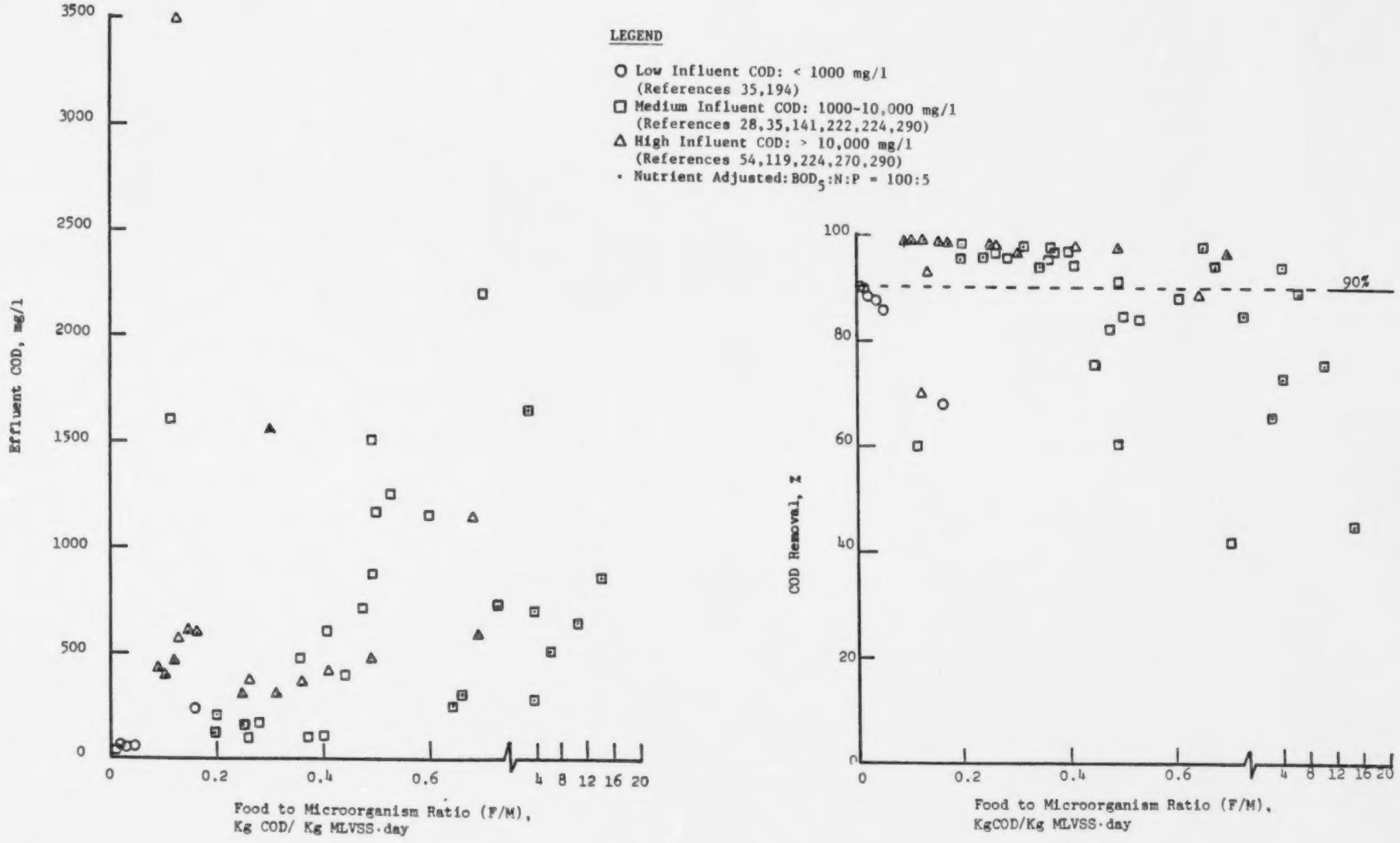

Figure 11. Relationship Between F/M Ratio and COD Removal for Bench-Scale Activated Sludge Studies

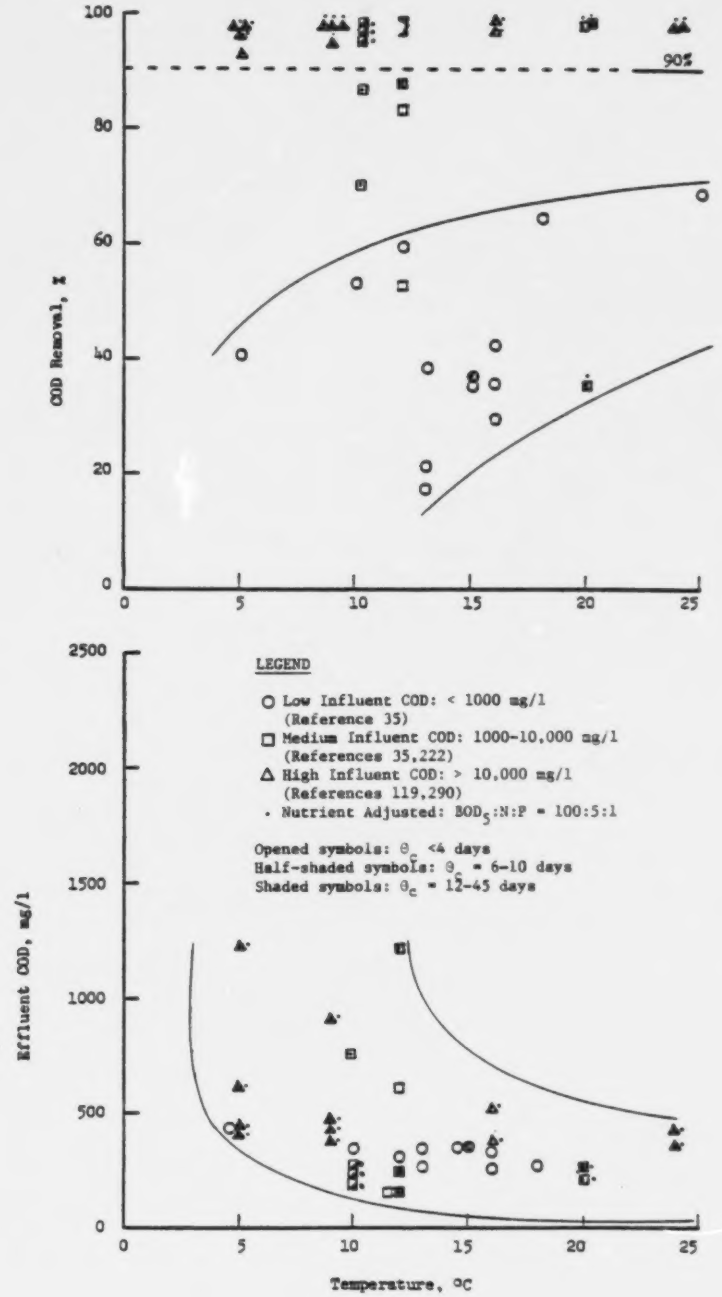

Figure 12. Relationship Between Temperature and COD Removal for Bench-Scale Activated Sludge Studies

TABLE 7. SUMMARY OF HEAVY METAL AND ALKALI AND ALKALINE EARTH METAL REMOVAL DATA FOR THE BENCH-SCALE ACTIVATED SLUDGE PROCESS

Heavy metals

	Cd	Cr	Fe	Pb	Mn	Ni	Zn
Influent							
Concentration Range, mg/l	0.04-0.4	0.1-1.9	240-2130	0.17-1.44	13-41	0.18-0.65	31-220
Removal Range, %	85-99	75-98	96-99	82-98	90-99	39-75	96-99
Average Removal, %	96	92	98	89	97	60	99

Alkali and Alkaline Earth Metals

	Ca	Mg	K	Na
Influent				
Concentration Range, mg/l	88-3780	35-660	200-1060	430-1350
Removal Range, %	64-99	3-90	8-46	0-35
Average Removal, %	90	52	27	16

Nitrification--Nitrification of leachates in the activated sludge process has been studied in depth by only one researcher (Johansen, 1975), although other investigators have provided TKN, ammonia, and nitrate data as shown in Appendix Table A-6. The effluent ammonia content of these leachates was typically 200 to 300 mg/l unless amended with nutrients. The general performance of bench-scale activated sludge systems with emphasis on nitrification is illustrated in Figure 13. Since nitrifying bacteria will typically have lower growth rates than carbonaceous bacteria, longer θ_c are needed for complete nitrification. As shown in Figure 13, θ_c of 10 days or longer were necessary to achieve better than 90% nitrification at temperatures above 12°C. At lower θ_c, viz., in the range of the limiting θ_c (6-10 days) for BOD_5, 60 to 80% nitrification was typically encountered.

Air stripping of NH_3 was believed to have occurred during one activated sludge study (Uloth and Mavinic, 1976). In this study, 96 to 99% of NH_3 was removed through aeration of the activated sludge units. The study was performed at 23°C and a pH of 8.5 to 8.8 and, although this pH was not as high as advocated for conventional stripping, the long detention times of 10 to 60 days and high NH_3 levels (1400-1800 mg/l) may have enhanced the effectiveness of ammonia removal by this mechanism.

Combined Treatment with Municipal Wastewater--The combination of industrial wastewaters with larger volumes of municipal wastewater has proven to be a successful treatment strategy on both bench- and pilot-scale (Table 8). Combined treatment may provide a better effluent as a result of the maintenance of a more heterogeneous population, the increased availability of nutrients, and the dilution of potential inhibitors.

TABLE 8. BENCH-SCALE RESEARCH PERFORMED ON COMBINED TREATMENT OF LEACHATE AND DOMESTIC WASTEWATER USING THE ACTIVATED SLUDGE PROCESS.

REFERENCE	PROCESS DESCRIPTION	PROCESS OBJECTIVE(S)	LEACHATE SOURCE
19,20	Semi-continuous, complete-mix, extended air reactor.	Determine optimum leachate to domestic wastewater ratio for organic removal; evaluate sludge production.	Landfill
42	Plug flow reactor for control and test unit operated at equivalent F/M.	Evaluate effect of adding 0.5% leachate to domestic wastewater for same F/M as domestic wastewater only case; characterize sludge settling characteristics.	Lysimeter and landfill
43	Plug flow reactor.	Study sequential uptake of different organic components.	Lysimeter
44,45,69, 70,286	Three complete-mix, continuous flow reactors in series to simulate a plug flow reactor.	Determine optimum leachate to domestic wastewater ratio for constant F/M, optimum F/M for constant ratio based on organic removal; characterize sludge settling properties.	Lysimeter and landfill
59	Complete-mix, continuous flow reactor.	Effect of shockloading of leachate to activated sludge process; evaluate sludge settling characteristics.	Synthetic leachate made of sodium acetate, acetic acid, glycine and pyrogallol
176,260, 261	Complete-mix, daily fill and draw reactor.	Determine optimum BOD_5:N:P ratio for various leachate to domestic wastewater ratios; characterize metal content in sludge.	Lysimeter

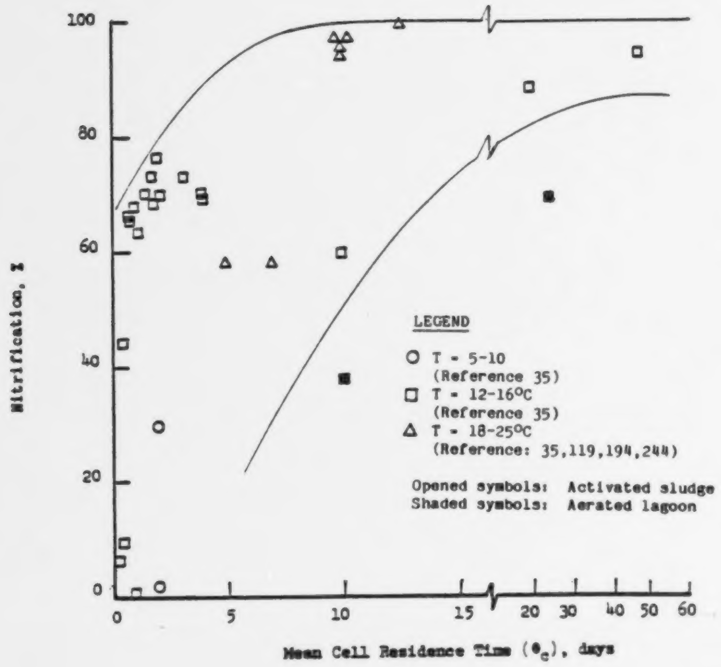

Figure 13. Relationship Between θ_c and Nitrification for Bench-Scale Activated Sludge Studies

Since only one pilot-scale study has been reviewed, pilot- and bench-scale studies were considered together. As indicated in Appendix Table A-7, a number of leachate to domestic wastewater volume ratios (L/D, expressed as % leachate) have been studied, resulting in a fairly broad range of organic influent strengths (150-3640 mg/l BOD_5).

Combined treatment of leachates was successful in removing 98 to 99% BOD_5 and 95% COD, although greater air requirements were generally reported. Increases as high as 400% and 800% in oxygen availability have been found necessary for successful treatment at 10% and 20% leachate to domestic wastewater (L/D) fractions, respectively, over the oxygen used in the domestic wastewater control (Boyle and Ham, 1972, 1974). Solids production was also higher, resulting in 300% and 800% more solids at 10% and 20% leachate to wastewater ratios, respectively. Moreover, sludge settleability was negatively affected by leachate introduction. Settling velocities for 1 to 3% L/D were determined to be about one-half of the control settling velocities by Chian and Dewalle (1976, 1977) and DeWalle and Chian (1977). Furthermore, excessive sludge bulking has been noted for L/D of 5,10, and 20% with a sludge volume index (SVI) of 100, 200 and 1000% higher than the control (Boyle and Ham, 1972, 1974).

Figure 14 provides an illustration of the relative success of combined leachate/wastewater treatment in terms of effluent organics (BOD_5, COD) and removal percentages. However, these data plots fail to reveal sludge handling difficulties and a consideration of them would lead to more conservative conclusions regarding the feasibility of this approach. Therefore, when sludge handling is considered, it appears that an L/D of less than 5% is required to apply this treatment strategy.

Aerated Lagoon--

Aerated lagoons are similar in many respects to activated sludge systems. Both processes utilize mechanical or diffused aeration to provide oxygen and mixing. Although not as widely practiced, aerated lagoons may also employ biomass recycle to increase cell retention time (θ_c) in a fashion similar to the activated sludge process. However, aerated lagoons are more typically operated as single-pass reactors with long hydraulic retention times.

Organic substrate (BOD_5, COD) removal was again utilized as the primary indicator of aerated lagoon process performance. The data presented in Figures 15 and 16 were segregated according to temperature. Most of the studies involved medium- to high-strength leachates, characterized by influent COD concentrations of 6400 to 9840 mg/l and BOD_5/COD ratios of 0.4 to 0.7 as indicated in Appendix Table A-8. Long retention times ranging from 7 to 100 days were used in the studies, and most were greater than 10 days. Therefore, the relationships between BOD_5 and COD removals and retention times (τ) for the aerated lagoon studies shown in Figure 15 were not particularly instructive. Temperature effects were also nondiscernible due to the long retention times.

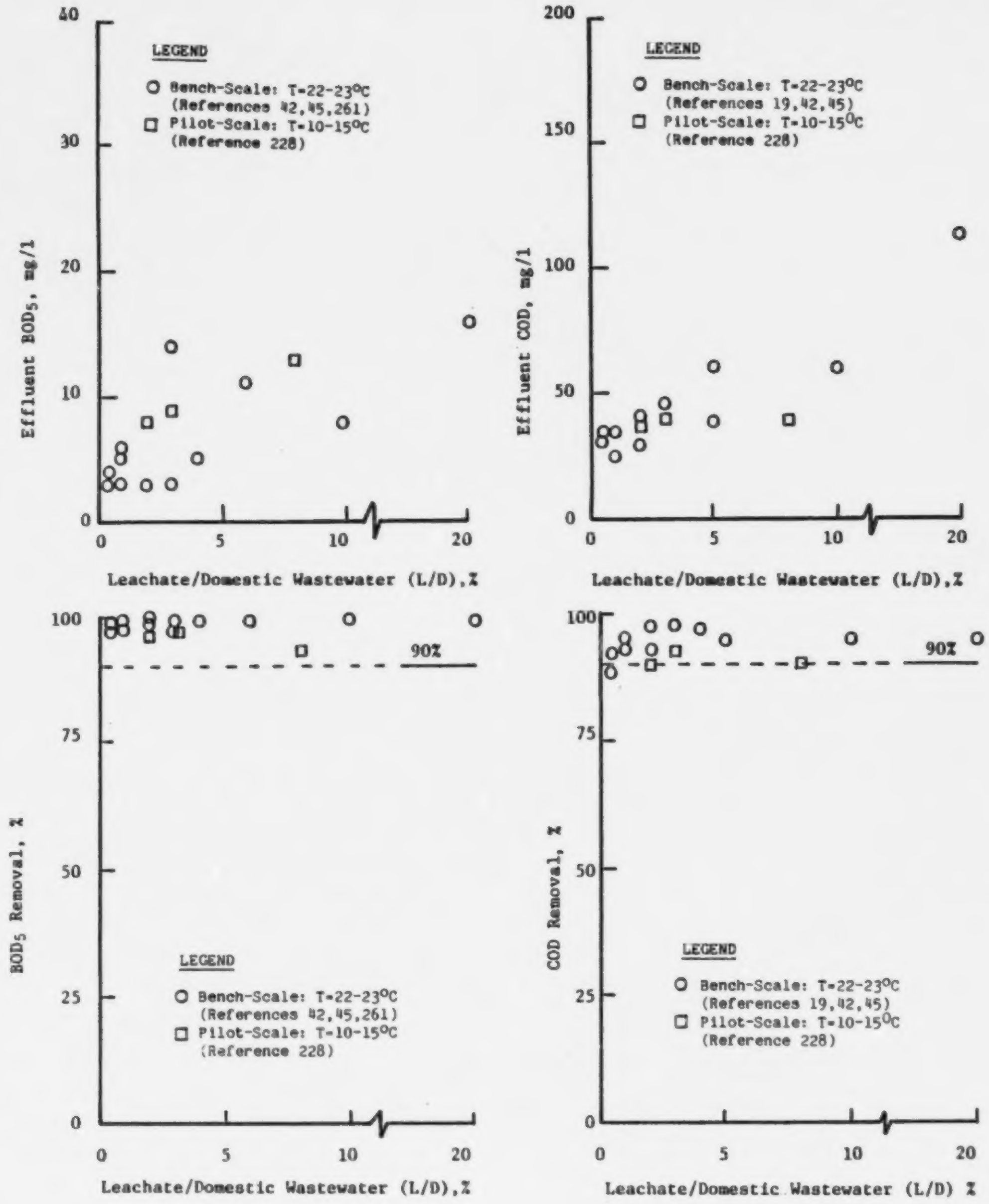

Figure 14. Relationship Between Leachate/Domestic Wastewater Volume Ratio and Organics (BOD_5, COD) Removal for Bench- and Pilot-Scale Combined Wastewater Activated Sludge Studies

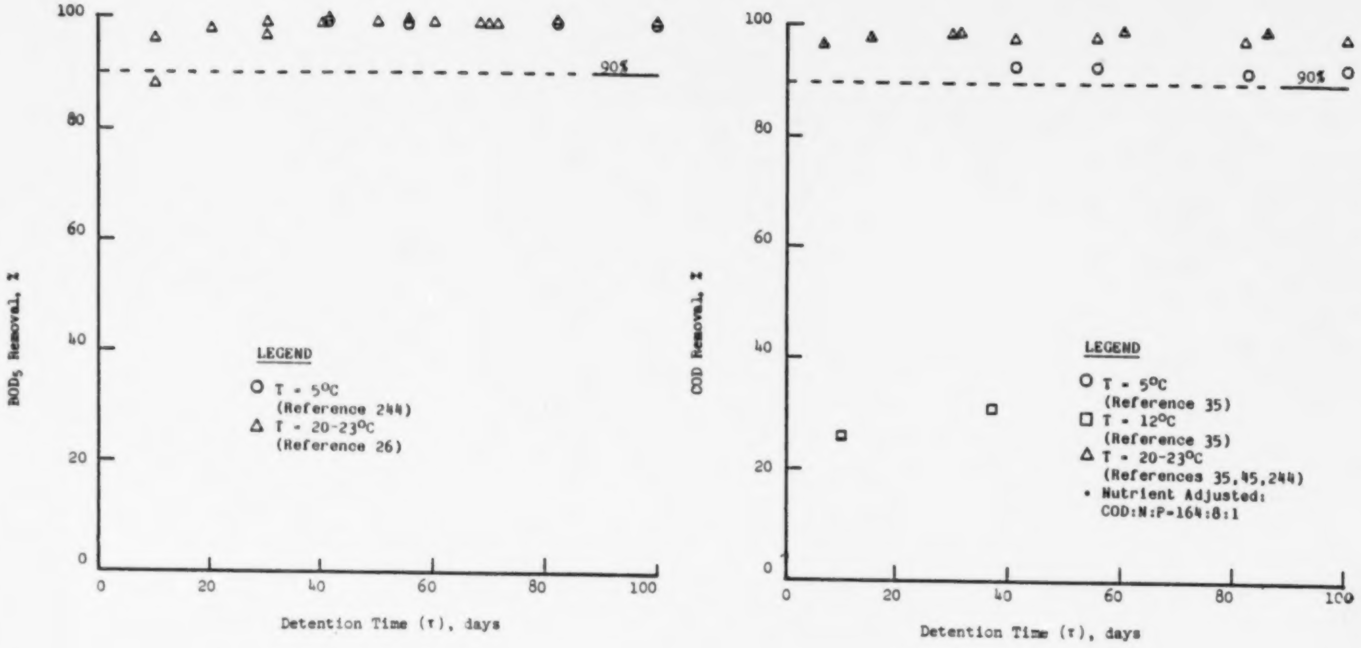

Figure 15. Relationship Between Hydraulic Retention Time (τ) and Organics Removal for Bench-Scale Aerated Lagoon Studies

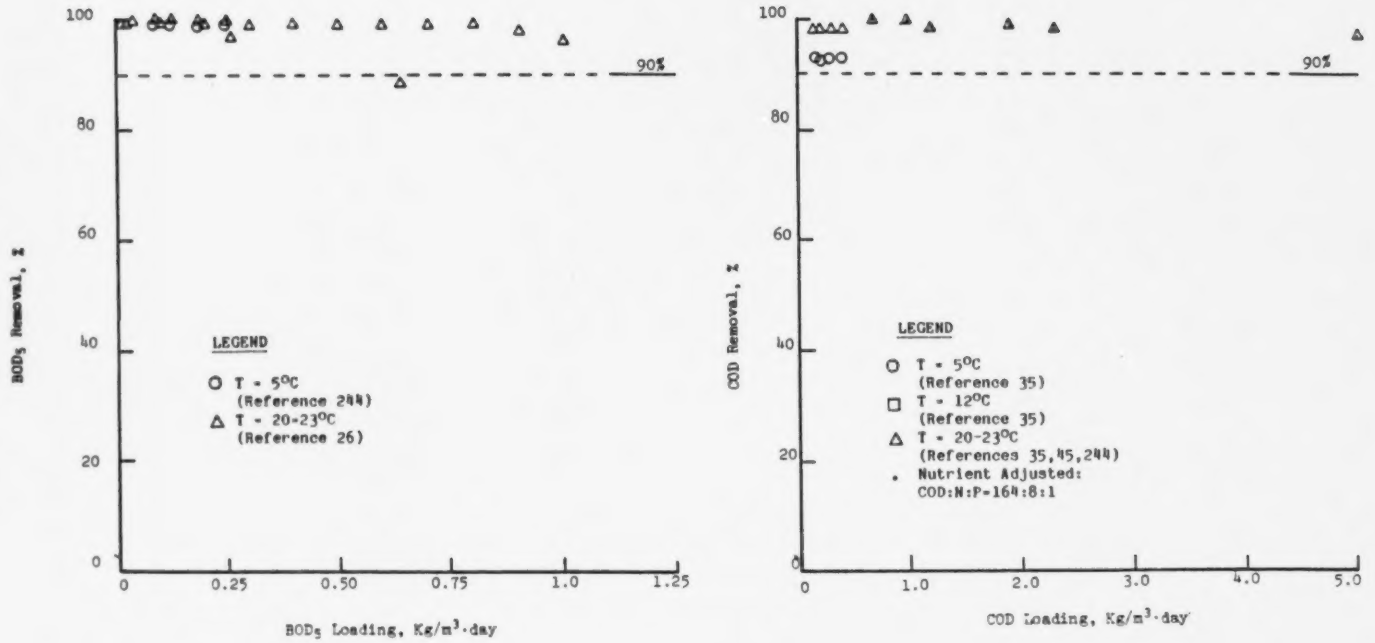

Figure 16. Relationship Between Organic Loading Rate and Organics Removal for Bench-Scale Aerated Lagoon Studies

Figure 16 provides an illustration of a similar lack of trend between organic loading and the performance of aerated lagoon processes. Loading rates as high as 1 kg $BOD_5/m^3 \cdot day$ or 5 kg $COD/m^3 \cdot day$ provided 90% or better BOD_5 and COD removal at 20 to 23°C.

Fixed-Film Processes--

The trickling filter and rotating biological contactor (RBC or biodisk) have been evaluated by only one investigator (Johansen, 1975). Unfavorable results were obtained in either case. However, a chemically precipitated leachate was used in the trickling filter study and also in one of two biodisk studies as indicated in Table 9. The raw leachate used in the other biodisk study was also a low-strength leachate characterized by a COD of 730 mg/l and COD/TOC ratio of 3.7. Retention times utilized were also very low. Therefore, this isolated study should probably not be construed as conclusive evidence that these processes are inapplicable. Given additional investigative evidence, these process options may also represent viable leachate treatment alternatives.

TABLE 9. EXPERIMENTAL CONDITIONS AND PERFORMANCE DURING TRICKLING FILTER AND ROTATING BIOLOGICAL CONTACTOR TREATMENT OF LEACHATE

	Trickling Filter	Rotating Biological Contactor	
Leachate*	CP*	CP	R
Influent BOD_5, mg/l	50	50	-
Influent COD, mg/l	380	400	730
Influent TOC, mg/l	114	114	200
BOD_5/COD	0.13	0.13	-
COD/TOC	3.3	3.5	3.7
BOD_5 Loading	0.1 $kg/m^3 \cdot day$	0.78 $g/m^2 \cdot day$	
COD Loading	0.9 $kg/m^3 \cdot day$	6.2 $g/m^2 \cdot day$	1.8 $g/m^2 \cdot day$
BOD_5 Removal, %	-	-	-
COD Removal, %	7.4	16	47
TOC Removal, %	7.5	24	44
BOD_5:N:P	100:200:0.2	100:200:0.2	-
pH	7.2-9.1	7.0-8.9	8.0-8.7
Temperature, °C	17	17	11
Reactor Type**	CSTR	PFR	PFR
Recycle Ratio	100	0	0
τ, hours	9	7	45

- = Data not given
*CP = Chemically pretreated
R = Raw leachate
**CSTR = Continuously-stirred tank reactor
PFR = Plug flow reactor

Kinetic Parameters for Bench-Scale Aerobic Processes--

Kinetic parameters associated with Monod-type expressions have been used to describe and understand microbial growth and substrate removal patterns associated with waste treatment processes. These kinetic parameters have been determined by a number of researchers from their experimental data as summarized in Table 10. The kinetic parameters of interest were cell yield (Y), decay coefficient (b), maximum specific growth rate (μ_{max}), and the saturation constant (K_s).

Similarities and dissimilarities existed when comparisons were made between parameter values for leachate and domestic wastewater, and for various influent substrate concentrations. The yield, Y, was fairly consistent, ranging from 0.29 to 0.59 mg VSS/mg BOD_5 or COD. However, the decay coefficient, b, was found to be variable, 0.002 to 0.336 day^{-1}, which might be attributed to inhibition by high NH_4^+, heavy metals or organic concentrations. Phosphorus limitation would also cause a higher decay rate. Maximum growth rates (μ_{max}) were also determined to be variable and to be both less than and greater than the typical values for domestic wastewater. Reported μ_{max} values for leachate treatment ranged from 0.02 to 16 day^{-1} based on BOD_5 or 0.3 to 24 day^{-1} based on COD, as compared to 1 to 8 day^{-1} and 4 to 11 day^{-1} for wastewater on BOD_5 and COD bases, respectively. The saturation constant, K_s, was the most variable parameter and was usually higher for leachates than typical values for domestic wastewater. This was partly attributed to the organic complexity and, therefore, more refractory nature of the leachate as a substrate.

Pilot- and Full-Scale Aerobic Treatment

Activated Sludge (AS), aerated lagoons (AL), and stabilization ponds (SP) have been investigated for the treatment of landfill leachates on pilot- and full-scale. A listing of pertinent literature citations for four activated sludge, four aerated lagoon, and four stabilization pond studies is provided in Table 11.

Activated Sludge--

As indicated in Table 11, the activated sludge process has been used on pilot- and full-scale for the treatment of leachate at four landfill sites. A summary of the leachates produced at each site and details of each AS configuration studied are presented in Table 12. The data in Table 12 have been separated into influent and effluent quality, pretreatment, treatment, post-treatment, and sludge characteristics sub-categories. Within the influent and effluent quality category, BOD_5 and COD data have been included to represent organic constituents, and iron has been included to reflect heavy metals behavior. Ammonia and TKN data were also included to evaluate the possible occurrence of nitrification.

The activated sludge processes summarized in Tables 11 and 12 have been fairly successful for the removal of organics and somewhat less successful for the removal of metals such as iron. In West Germany, 94 to 98% BOD_5 removal was consistently achieved at a θ_c of 12 days, even at temperatures as low as 6 to 7°C (Scherb, 1981). The facilities at Bucks County, PA (Steiner, et al., 1977 a,b, 1979, 1980; Stoll, 1979) have tested a number of operating strategies including NH_3 stripping as a pretreatment measure, the use of two tanks in series or parallel operation, and nutrient additions. The results of

TABLE 10. SUMMARY OF MONOD KINETIC PARAMETERS FOR ACTIVATED SLUDGE TREATMENT OF LEACHATE

REFERENCE	INFLUENT CONCENTRATION, mg/l		Y, $\frac{\text{mgVSS}}{\text{mg BOD}_5\text{ or COD}}$		b, day^{-1}		μ_{max}, $\frac{\text{mgBOD}_5\text{ or COD}}{\text{mgVSS}\cdot\text{day}}$		K_s, mg/l		T, °C
	BOD$_5$	COD	BOD$_5$	COD	BOD$_5$	COD	BOD$_5$	COD	BOD$_5$	COD	
28	-	9760	-	0.35	-	0.084	-	0.28	-	673	22-24
45*	-	35,000- 58,000	-	0.42	-	0.025	-	-	-	-	22-24
53,54,97	7100	15,800	-	0.4	-	0.05	-	0.6	-	175	22-24
118*	12,900	19,400	0.49	0.34	0.015	0.016	0.15	0.5	12.3	1800	5
151*	-	2400-4500	-	0.29**	-	2.4	-	24	-	1460	22-24
193,194	230	360	-	0.59	-	0.115	-	1.06	-	182	22-24
205,207	260	500	0.50	-	0.336	-	16	-	41.3	-	22-24
269,270*	36,000	48,000	0.332	-	0.0025	-	0.25	-	21,380	-	22-24
284*	8090	13,000	0.49	-	0.009	-	0.57	-	82	-	22-24
	8090	13,000	0.51	-	0.018	-	0.57	-	64	-	15
	8090	13,000	0.51	-	0.006	-	0.26	-	35	-	10
	8090	13,000	0.55	-	0.002	-	0.19	-	17	-	5
288*	13,600	19,300	0.374	-	0.015	-	0.28	-	19.6		22-24
168	1000	1700	0.59	0.42	0.40	0.56	4.5	-	99	-	21-24
Domestic Wastewater, Metcalf and Eddy, 1979			0.4-0.8	0.35-0.45	0.04-0.075	0.05-0.10	1-8	4-11	25-100	15-70	22-24

*Nutrient adjusted; BOD$_5$:N:P = 100:5:1

**Based on dehydrogenase activity rather than VSS measurement as viable organism concentration.

TABLE 11. LANDFILLS WITH PILOT- OR FULL-SCALE AEROBIC LEACHATE TREATMENT FACILITIES

REFERENCE	SCALE	PROCESS*	PROCESS DESCRIPTION	LANDFILL LOCATION
15	Full	AL	Lime neutralization as pretreatment prior to biological treatment with effluent discharge to surface water.	Allegheny County, Pennsylvania (2 landfills)
26	Full	AL	Lime addition for metal removal, neutralization prior to biological treatment with effluent discharge to POTW.	North Hempstead, New York
113,114,115	Full	AL	Facultative aerated lagoon with land disposal of treated effluent.	Jefferson County, Missouri
157	Full	AL,SP	Series of four ponds, flow equalization, aeration, and two stabilization ponds with discharge to surface water for effluent disposal.	West Germany
166	Full	SP	Four ponds operated at very long detention time followed by effluent discharge to surface waters. Use of aquatic plants to enhance treatment performance.	Barre, Massachusetts
187	Full	AS,SP	Aeration tank with clarifier operated without solids recycle followed by spray irrigation of effluent. Leachate storage in stablization pond for winter.	England
228	Pilot	AS	Complete-mix aeration tank with clarifier operated with solids recycle.	West Germany
231	Full	AS,SP	Chemical addition as pretreatment prior to activated sludge or stabilization pond treatment with effluent discharge to surface water.	Pennsylvania (2 landfills)
244	Full	AL	Series of five diffused aeration lagoons and one settling lagoon followed by spray irrigation of effluent flow equalization.	West Germany
245,246,247 248,256	Full	AS	Lime addition and ammonia stripping and neutralization followed by biological treatment operated on no solids recycle basis for organic substrate removal and nitrification. Effluent disposal by spray irrigation or surface water discharge.	Bucks County, Pennsylvania

*AS = Activated Sludge
AL = Aerated Lagoon
SP = Stabilization Pond

series and parallel operations were fairly similar, with both systems achieving 92 to 97% BOD_5 removal at loading rates of 1.5 to 1.8 kg $BOD_5/m^3 \cdot day$ and θ_c of 2 to 4 days. On a yearly basis, the series mode provided superior nitrification; nitrification efficiencies for the parallel reactors decreased from 95% at 15 to 29°C to 40% at 0 to 12°C.

In other studies (Klingl, 1981), a higher-strength leachate was used to provide a BOD_5 loading of 6.3 $kg/m^3 \cdot day$. Organic removal efficiency with this loading was somewhat lower at 83 to 94% BOD_5 removal and 78 to 89% COD removal. It was reported that process inhibition was attributable to NH_3, although the low retention time and high loading rate were probable contributors as well.

TABLE 12. SUMMARY OF LEACHATE TREATMENT PERFORMANCE AND DESIGN PARAMETERS FOR PILOT-SCALE AND FULL-SCALE ACTIVATED SLUDGE TREATMENT FACILITIES

REFERENCE / ITEM	(187)	(228)		(231)	(245,246,247,248,256)			
Treatment Scale	Full	Pilot	Pilot	Full	Full S1-P*	Full S1-P*	Full S3*	Full S4*
Influent Quality								
BOD_5, mg/l	1340	3580	3580	-	3560	5970	12,600	12,600
COD, mg/l	2460	4540	4540	2500	6480	9870	21,200	21,200
TOC, mg/l	-	1710	1710	-	-	-	-	-
BOD_5/COD	0.54	0.79	0.79	-	0.55	0.60	0.59	0.59
COD/TOC	-	2.7	2.7	-	-	-	-	-
TKN, mg/l	-	-	-	90	290	-	708	708
NH_3-N, mg/l	168	-	-	50	290	740	649	649
BOD_5:N:P	100:8:-	100:5:0.5	100:5:0.5	-	100:5:1	100:5:1	100:5:0.02	100:5:0.02
Fe, mg/l	10	-	-	750	3.6	1.9	350	350
pH	-	-	-	6.0	7.6	7.6	7.6	7.6
Temperature, °C	12-18	6-7	10-15	0-25	0-24	20-24	20-29	20-29
Q, m^3/day	150	1.17	1.17	662	79.5	37.9	75.7	75.7
Pretreatment	No	No					No	No
Chemical addition			No	-	Lime, H_3PO_4, H_2SO_4	Lime, H_3PO_4, H_2SO_4		
Flow equalization		No	No	No	No	Yes		
Nutrient addition		Yes	Yes	No	Yes	Yes		
NH_3 stripping		No	No	No	Yes	Yes		
Primary clarification		No	No	No	No	No		
Treatment								
Aeration								
BOD_5 loading, kg/m^3·day	1.3	0.30	0.30	-	1.8	1.5	6.3	6.3
COD loading, kg/m^3·day	2.5	0.39	0.39	-	3.2	2.5	11	11
MLVSS, g/l	-	4-5	4-5	-	6-12	6-12	6-12	6-12
F/M, BOD_5/MLVSS·day	-	0.16	0.16	-	0.30-0.15	0.25-0.12	0.53-0.26	0.53-0.26
F/M, COD/MLVSS·day	-	0.20	0.20	-	0.56-0.27	0.42-0.20	0.90-0.44	0.90-0.44
τ, hours	24	144	144	-	48	96	48	48
Θ_c, days	1	12	12	-	2	4	2	2
Air, m^3/min	-	-	-	-	14.2	14.2	14.2	14.2

TABLE 12 (Continued)

Secondary Clarification								
Overflow rate, $m^3/m^2\cdot day$	2	0.7-1	0.7-1	-	1.5	1.4	1.4	1.4
τ, hours	48	24-40	24-40	-	14	15	15	15
Posttreatment	No	No	No					
Chemical addition				No	No	No	No	No
Sand filtration				Yes	No	No	No	No
Chlorination				No	Yes	Yes	Yes	Yes
Effluent Quality								
BOD_5, mg/l	1215	220	40	-	120	460	760	2150
COD, mg/l	2280	380	200	-	940	1090	2260	4680
BOD_5 removal, %	9.3	94.0	97.8	-	96.7	92.2	94.0	83.0
COD removal, %	7.3	91.6	95.6	-	85.5	89.0	89.3	77.9
TKN, mg/l	-	-	-	-	102	-	153	312
NH_3-N, mg/l	164	-	-	-	80	6.3	76.4	51.9
NH_3-N removal, %	2	-	70	-	72.0	99.1	76.4	51.9
Fe, mg/l	-	-	-	-	3.0	0.7	1	200
Fe removal, %	-	-	-	-	17	62	99.7	44
pH	-	-	-	-	7.6	7.6	10.2	8.6
Sludge Characteristics								
Solids, %	-	-	1.3	-	-	-	-	-
Volatile, %	-	-	50	-	-	-	-	-
SVI, ml/g	-	<300	50	-	80	80	-	-
Effluent Disposal	Spray irrigation, 9.4 $l/m^2\cdot day$, 60% BOD_5 removal, 35% COD removal, 60% NH_3-N removal	Surface water	Surface water	Surface water	Surface water or spray irrigation			

-Data not given
*S1-P: System 1 with aeration tanks in parallel
S1-S: System 1 with aeration tanks in series
S3-P: System 3 with aeration tanks in parallel
S4-P: System 4 with aeration tanks in parallel

The English facility (Newton, 1979) did not perform well with regard to either carbon or ammonia removal. The very low cell retention time (1 day) probably did not allow sufficient time for effective substrate removal.

Aerated Lagoon--

The aerated lagoon has been the most commonly used process for leachate treatment on full-scale. Treatment performance and design parameters for the full-scale aerated lagoon treatment facilities reviewed in this study are presented in Table 13. Although five facilities were reported, only three could be evaluated since sufficient information was not available for two of the facilities (Brownell, *et al*., 1982; Goeppner, 1975 a,b) Overall, the aerated lagoon proved to be an effective means for leachate treatment in terms of BOD_5, COD, and Fe removal. Detention times ranging from 7 to 135 days provided 70 to 99% BOD_5 removal and 70 to 95% COD removal. Typically, 90% BOD_5 removal was achieved along with 92 to 99% Fe removal.

Data on the treatment of various leachates which were characterized by influent BOD_5 concentration and the BOD_5/COD ratio were also provided by Stegmann (1981). It appears evident from these data (Table 13) that leachates with a low BOD_5/COD ratio (0.05-0.2) required very long detention times to achieve substantial BOD_5 removal. This might be attributed to the resistance of humic and fulvic acid substances that result during organic substrate assimilation which are less biodegradable than the original organic substrate (Chian and DeWalle, 1977a,b).

Stabilization Pond--

The stabilization pond has been used at four existing landfills reported in the literature. A summary of the treatment performance and design parameters for the full-scale stabilization ponds is given in Table 14. In spite of its simplicity, the stabilization pond generally achieved satisfactory treatment performance with 93 to 99% BOD_5 removal and 90 to 99% COD removal at τ = 60 to 90 days. However, low BOD_5 and COD removals were reported by Klingl (1981) for low-strength leachates as characterized by low influent BOD_5 and COD concentrations and low BOD_5/COD ratios (0.06 - 0.08). High Fe removal (91-99%) was achieved for τ = 60 to 90 days. NH_3-N removal varied from 22 to 99% removal for τ = 43 to 90 days. BOD_5, COD, NH_3-N and Fe removals improved as τ was decreased.

Treatment and Disposal of Aerobic Process Solids

Solids treatment and disposal are essential aspects of all aerobic biological treatment processes. Therefore, sludge solids characterization is helpful in designing and evaluating the operation of sludge treatment units and determining acceptable methods for final disposal. Seven parameters have been used to characterize such solids as listed below along with their intended uses.

TABLE 13. SUMMARY OF LEACHATE TREATMENT PERFORMANCE AND DESIGN PARAMETERS FOR FULL-SCALE AERATED LAGOON FACILITIES.

REFERENCE / ITEM	(15)		(26)	(113-115)	(157)	(244)			
Influent Quality									
BOD_5, mg/l	120	3000	10,000	800	5310	4500	3000	650	100
COD, mg/l	-	-	14,000	1500	7800	-	-	-	-
TOC, mg/l	-	-	-	455	2740	-	-	-	-
BOD_5/COD	-	-	0.71	0.53	0.68	>0.4	0.4	0.2	0.05
COD/TOC	-	-	-	3.3	2.8	-	-	-	-
TKN, mg/l	-	-	700	-	270	-	-	-	-
NH_3-N, mg/l	10	30	600	-	240	-	-	-	-
BOD_5:N:P	100:8:-	100:1:-	100:5:1	-	100:5:0.4	-	-	-	-
Fe, mg/l	60	120	700	-	60	-	-	-	-
pH	6.8	5.6	-	6.0	6.8	-	-	-	-
Temperature, °C	10-25	10-25	2-25	0-25	0-25	5-20	5-20	5-20	5-20
Q, m^3/day	355	45.4	303	37.9	216	-	-	-	-
Pretreatment				No		No	No	No	No
Flow equalization	No	No	No		τ=2d				
Lime addition	Yes	Yes	Yes		No				
Settling	τ=2d	τ=2d	Yes		No				
Neutralization	No	No	Yes		No				
Nutrient addition	No	No	Yes		Yes				
Preaeration	No	No	Yes		No				
Treatment									
BOD_5 loading, kg/m^3·day	0.01	-	0.33	0.01	0.76	0.03	0.03	0.005	0.001
COD loading, kg/m^3·day	-	-	0.47	0.02	1.1	-	-	-	-
τ, days	20	-	30	90	7	135	120	120	90
Post-treatment				No		No	No	No	No
Settling	τ=2d	τ=2d	No		τ=5.3 hrs				
Chlorination	Yes	No	No		No				
Chemical addition	No	No	H_2O_2		No				
Effluent Quality									
BOD_5, mg/l	10	920	-	-	50	25	25	25	25
COD, mg/l	-	-	-	-	415	-	-	-	-
BOD_5 removal, %	92	70	-	-	99	>99	>99	96	75
COD removal, %	70	-	-	-	95	-	-	-	-
TKN, mg/l	-	-	-	-	100	-	-	-	-
NH_3-N, mg/l	-	-	-	-	80	-	-	-	-
NH_3-N removal, %	-	-	-	-	66	-	-	-	-
Fe, mg/l	1	10	-	-	0.2	-	-	-	-
Fe removal, %	98	92	-	-	>99	-	-	-	-
pH	7.5	7.4	-	-	8.1	-	-	-	-
Effluent Disposal	Surface water for both		POTW	Ridge and furrow land disposal, rate: 4.71/m^2·day	Stabilization pond	Spray irrigation for all four cases			

TABLE 14. SUMMARY OF LEACHATE TREATMENT PERFORMANCE AND DESIGN PARAMETERS FOR FULL-SCALE STABILIZATION POND FACILITIES

REFERENCE / ITEM	(157)		(166)	(187)	(230)
Influent Quality					
BOD_5, mg/l	28	21	21,100	1340	-
COD, mg/l	370	330	35,700	2460	2500
TOC, mg/l	92	74	-	-	-
BOD_5/COD	0.08	0.06	0.59	0.54	-
COD/TOC	4.0	4.5	-	-	-
TKN, mg/l	92	76	-	-	90
NH_3-N, mg/l	81	63	440	168	50
BOD_5:N:P	100:290:5	100:300:10	100:2:-	100:8:-	-
Fe, mg/l	-	-	1400	-	750
pH	8.4	8.5	5.2	-	6.0
Temperature, °C	18-23	18-23	0-25	0-18	0-25
Q, m^3/day	77.8	77.8	4	150	549
Pretreatment			No	No	
Chemical Addition	No	No			Yes
Biological Treatment	Yes	Yes			No
Settling	Yes	Yes			Yes
Nutrient Addition	No	No			No
Treatment					
BOD_5 loading, kg/m^3·day	<0.01	<0.01	0.70;0.35;0.23	0.01	-
COD loading, kg/m^3·day	<0.01	<0.01	1.2;0.60;0.40	0.03	-
τ, days	63	44	30;60;90	90	-
Post-treatment	No	No	No	No	No
Effluent Quality					
BOD_5, mg/l	21	13	4650;220;10	100	-
COD, mg/l	330	160	9500;400;120	-	-
BOD_5 removal, %	25	38	77;99;>99	93	-
COD removal, %	11	52	73;99;>99	-	-
TKN, mg/l	76	20	-	-	-
NH_3-N, mg/l	63	12	130;70;3.5	-	-
NH_3-N removal, %	22	81	70;84;99	-	-
Fe, mg/l	-	-	320;120;1.0	-	-
Fe removal, %	-	-	77;91;>99	-	-
pH	8.5	8.3	6.5;6.8;7.3	-	-
Effluent Disposal	Stab. Pond	Surface Water	Surface Water	Surface Water	Surface Water

- Data not given

Test	Use
· % Solids	Design of sedimentation units; operational parameter for sludge settleability.
· % MLVSS/MLSS	Design of sludge digestion units.
· Specific resistance	Design of sludge dewatering units.
· Filter yield	Design of sludge dewatering units; quantity for disposal.
· Settling velocity	Design of sedimentation units.
· Metal content	Determine acceptance for ultimate disposal.
· Sludge Volume Index (SVI)	Operational parameter to determine activated sludge settleability in sedimentation units.

Results of solids characterization tests are presented and compared to typical values reported for domestic wastewater activated sludge in Table 15. Percent solids for leachate-derived sludges (1.1-5.0%) was determined to be slightly greater than the typical value for domestic wastewater (0.5-1.5%). This was possibly attributable to higher inorganic content in the leachate sludges, especially in terms of iron and calcium. Moreover, the percent volatility expressed as % MLVSS/MLSS for the leachate sludge was slightly lower than for domestic wastewater sludge.

A significant difference was also noted between specific resistance values of typical domestic wastewater sludges and those reported for leachate-derived sludges. The specific resistance of leachate sludge without chemical conditioning (10^{12} m/kg) was reported as one to two orders of magnitude less than the wastewater sludge values (10^{13}-10^{14} m/kg) without chemical conditioning. However, when the sludge was preconditioned with a chemical or polymer, specific resistance for the leachate sludge was superior to the typical wastewater range; a difference of two to four orders of magnitude in specific resistance existed between the two sludges. The leachate activated sludge exhibited very good dewatering properties as indicated by the reported specific resistance values.

Filter yields, expressed as $kg/m^2 \cdot hr$, were similar between the two cases; the leachate sludge filter yield varied from 2.2 to 28 and the domestic wastewater activated sludge varied from 2.4 to 20 with chemical conditioning. The higher values indicated for the leachate sludge were induced by high chemical additions for conditioning. Settling velocities for the two types of sludges were also fairly similar, although the leachate involved a somewhat thicker suspension.

Iron (Fe) and zinc (Zn) were found to be present in high concentrations (75,000 mg Fe/kg SS and 4000 mg Zn/kg SS) in leachate sludges. Cadmium (Cd), chromium (Cr), lead (Pb), and manganese (Mn) levels in the leachate sludge were reported to be similar to those found in sewage sludges.

TABLE 15. SUMMARY OF SLUDGE CHARACTERISTICS FOR AEROBIC LEACHATE TREATMENT PROCESSES

REFERENCE	PROCESS*	SOLIDS,%**	$\frac{\text{MLVSS}}{\text{MLSS}}$,%	SPECIFIC RESISTANCE, m/kg	FILTER YIELD,$kg/m^2 \cdot hr$	SETTLING VELOCITY, cm/sec	METAL CONTENT IN WET SLUDGE, $\frac{\text{mg metal}}{\text{kg SS}}$ Cd	Cr	Fe	Pb	Mn	Zn
45	AL	-	-	1.4×10^{12} @ P = 37 cm Hg	2.2 @ -47 cm Hg	-	-	-	-	-	-	-
	AL	-	-	$0.07\text{-}1.3 \times 10^{12}$ w/$FeCl_3$	3.4-28 @ -47 cm Hg	-	-	-	-	-	-	-
	AL	-	-	$0.05\text{-}1.4 \times 10^{12}$ w/Lime	2.2-15 @ -47 cm Hg	-	-	-	-	-	-	-
	AL	-	-	$0.06\text{-}0.5 \times 10^{12}$ w/Polymer	5.4-12 @ -47 cm Hg	-	-	-	-	-	-	-
	AL	-	-	$0.2\text{-}0.4 \times 10^{12}$ w/Polymer	4.9-6.8 @ -47 cm Hg	-	-	-	-	-	-	-
	AL	2-4	-	-	-	0.001-0.02 @ 20-40 g/ℓ	-	-	-	-	-	-
53,54,97	AS	-	66-73	-	-	-	-	-	-	-	-	-
118,119	AS	-	47-64	-	-	-	-	-	-	-	-	-
151	AS	-	41-51	-	-	-	-	-	-	-	-	-
143	AS	1.1-5.0 (2.1)	22-64	$0.85\text{-}9.6 \times 10^{12}$	-	-	-	-	-	-	-	-
260,261	AS	-	59-63	-	-	-	5.3-7.6	47-168	77,000-89,000	17-127	2800	4400-5200
288-291	AS	-	53-67	-	-	-	-	-	-	-	-	-
Typical for Domestic Wastewater***	AS	0.5-1.5	50-80	4.8×10^{13}-2.8×10^{14}	2.4-20 with chemical conditioning	0.005-0.13 @ 1-14 g/ℓ	nd-1100 (87)	22-30,000 (1800)	<1000-40,000 (10,000)	80-26,000 (1900)	100-8800 (1200)	51-28,360 (3500)

- Data not given
* AS = Activated Sludge
AL = Aerated Lagoon
** After 30 min to 1 hour settling
*** References: Dick and Ewing (1967); Javaheri and Dick (1969); Karr (1976); Coackley (1960); Gale (1968); Dahlstrom and Cornell (1958); FPA (1976); Metcalf & Eddy (1979)

nd = not detected
( ) = mean value

The Sludge Volume Index (SVI) is commonly determined to evaluate sludge settleability, although its transferability between studies has definite recognized limitations. The test was originally designed for use in evaluating operational problems during settling of activated sludge. Despite its limitations, the SVI was examined for its potential as a relative indicator of sludge settleability among the data that were presented in the literature. Since SVI is a function of the suspended solids concentration, the data were segregated on the basis of MLSS concentrations. The SVI was then plotted versus θ_c to provide a relative indication of sludge settleability as shown in Figure 17. Overall, the SVI was frequently <75, possibly indicative of good sludge settleability.

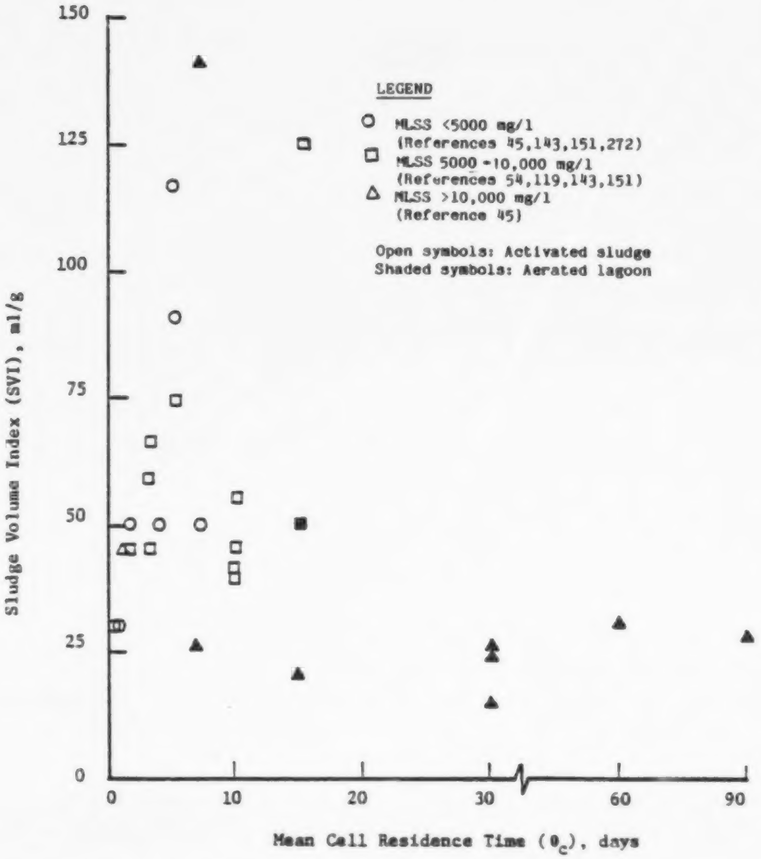

Figure 17. Relationship Between Sludge Volume Index and Mean Cell Residence Times for Aerobic Biological Treatment Studies

Although variations in sludge characteristics hamper the use of SVI as a universally accepted criterion, it is generally accepted that a SVI >100 reflects relatively poor settleability, whereas SVI <100 reflects relatively good settleability. On this basis, poor, good and very good sludge settleabilities were reported in the literature which indicated correspondingly variable sludge settling characteristics. In addition, deflocculation was reported to have occurred regardless of the degree of settleability. In fact, deflocculation sometimes occurred in sludges that were described as portraying very good settleability after the sludge had settled (Johansen, 1975). Pinpoint floc and sludge bulking were also identified for activated sludge processes treating leachate (Boyle and Ham, 1972, 1974; Graham and Mavinic, 1979; Graham, 1981).

ANAEROBIC BIOLOGICAL PROCESSES

Anaerobic biological treatment methods can provide a number of advantages over the traditional aerobic processes reviewed. In particular, an energy surplus may be available from the production of methane. Moreover, anaerobic cell yields are lower, resulting in lower sludge production and associated handling costs. Accordingly, the variables of interest in evaluating the feasibility of anaerobic treatment of leachate include methane production rates as well as the variety of indices used in describing aerobic treatment process performance.

Three general types of anaerobic treatment processes have been evaluated for the treatment of landfill leachates. These include external treatment in suspended-growth (SG) or attached-growth (AG) reactors, and *in situ* treatment using leachate collection and recycle back through the landfill. Little information was available beyond bench-scale for the external treatment systems. Therefore, these are discussed first, followed by a review of leachate recycle studies on all scales.

Bench-Scale Anaerobic Processes

The external anaerobic treatment strategies (SG, AG) studied on bench-scale (and one pilot-scale study) include applications of both completely-mixed and plug-flow reactors as summarized in Table 16. The experimental data associated with these studies are summarized in Appendix Table A-9 and utilized in the ensuing discussions of process variables.

Effect of Mean Cell Residence Time (θ_c)--

The data for bench-scale anaerobic treatment of leachates (Appendix Table A-9) were segregated as before on the basis of influent strength and biodegradability ratios. Limited BOD_5 data were available, therefore, only two influent categories (medium- and high-strength) were used in describing the effects of θ_c. Medium- and high-strength influents are described as having 1,000 to 5,000 mg/l BOD_5 or 1,000 to 10,000 mg/l COD and >5,000 mg/l BOD_5 or >10,000 mg/l COD, respectively. Leachates which received nutrient amendments are also distinguished.

The relationship between θ_c and organic removals is illustrated in Figures 18 and 19 for BOD_5 and COD, respectively. The data presented in these figures are from mesophilic (33-37°C) studies; studies at lower temperatures

TABLE 16. BENCH-SCALE ANAEROBIC BIOLOGICAL TREATMENT OF LEACHATE

REFERENCE	PROCESS	PROCESS DESCRIPTION	RESEARCH OBJECTIVE(S)	LEACHATE SOURCE*
14	Suspended growth	Plug flow, continuous upflow reactor. **	Effect of temperature on COD and metal removal efficiencies and gas production.	Landfill
19,20,147	Suspended growth	Plug flow, daily fill and draw reactor operation.	Effect of θ_c and BOD_5 and COD loading on BOD_5 and COD removal and gas production.	Landfill
33,176,217	Suspended growth	Complete-mix, continuous flow reactor.	Effect of BOD_5 and COD loading on BOD_5 and COD removal and gas production; determine kinetic parameters; extent of heavy metal removal.	Lysimeter
44,45,46	Attached growth	Complete-mix, continuous upflow filter containing plastic media.	Effect of pH adjustment, sludge seeding, shockloading and θ_c on COD and metal removal efficiencies and gas production.	Lysimeter
68,70,73	Attached growth	Complete-mix, continuous upflow filter containing plastic media.	Determine operating variable that controls heavy metal removal efficiency.	Landfill
98,220	Attached growth; Suspended growth	Plug flow, continuous upflow filter containing crushed limestone as contact media; complete-mix, hourly fill and draw reactor operation.	Effect of temperature, pH adjustment, nutrient addition, and COD loading on COD and TOC removal efficiencies; determine kinetic parameters.	Landfill
135,136	Suspended growth	Complete-mix continuous flow reactor.	Effect of θ_c on BOD_5 and COD removal efficiencies and gas production; effect of sodium inhibition.	Landfill
143	Suspended growth	Plug flow, upflow filter containing crushed limestone as surface contact media, operated fill and draw and continuous flow mode.	Effect of BOD_5 and COD loading, temperature, and effluent recirculation on BOD_5, COD and metal removal efficiencies and gas production.	Landfill
151,205, 206,207	Suspended growth	Complete-mix, continuous flow reactor.	Effect of θ_c on BOD_5 and COD removal efficiencies; determine kinetic parameters.	Landfill and lysimeter
223	Attached growth	Plug Flow, upflow filter containing rock media.	Effect of θ_c on organics removals and gas production.	Landfill
30	Suspended growth	Complete-mix, semi-continuous flow reactor.	Effect of feed concentration on organics removal.	Landfill
237	Attached growth	Plug flow, upflow filter containing plastic media.	Effect of HRT on organics, metal removals and gas production.	Landfill

*All leachate sources are characteristic of municipal solid waste landfills.
**Pilot-scale study.

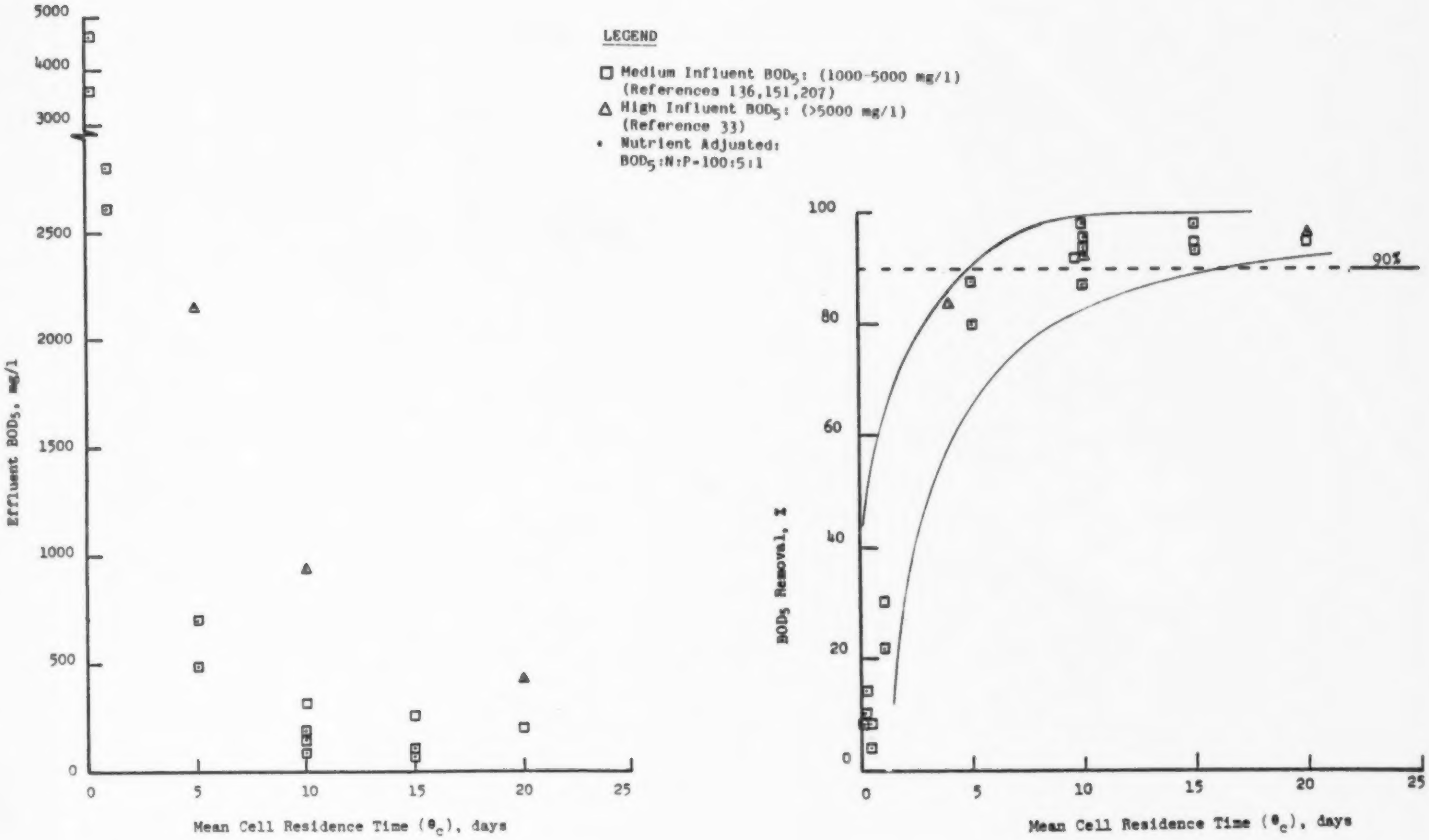

Figure 18. Relationship Between Mean Cell Residence Time and BOD_5 Removal for Bench-Scale Anaerobic Treatment Studies

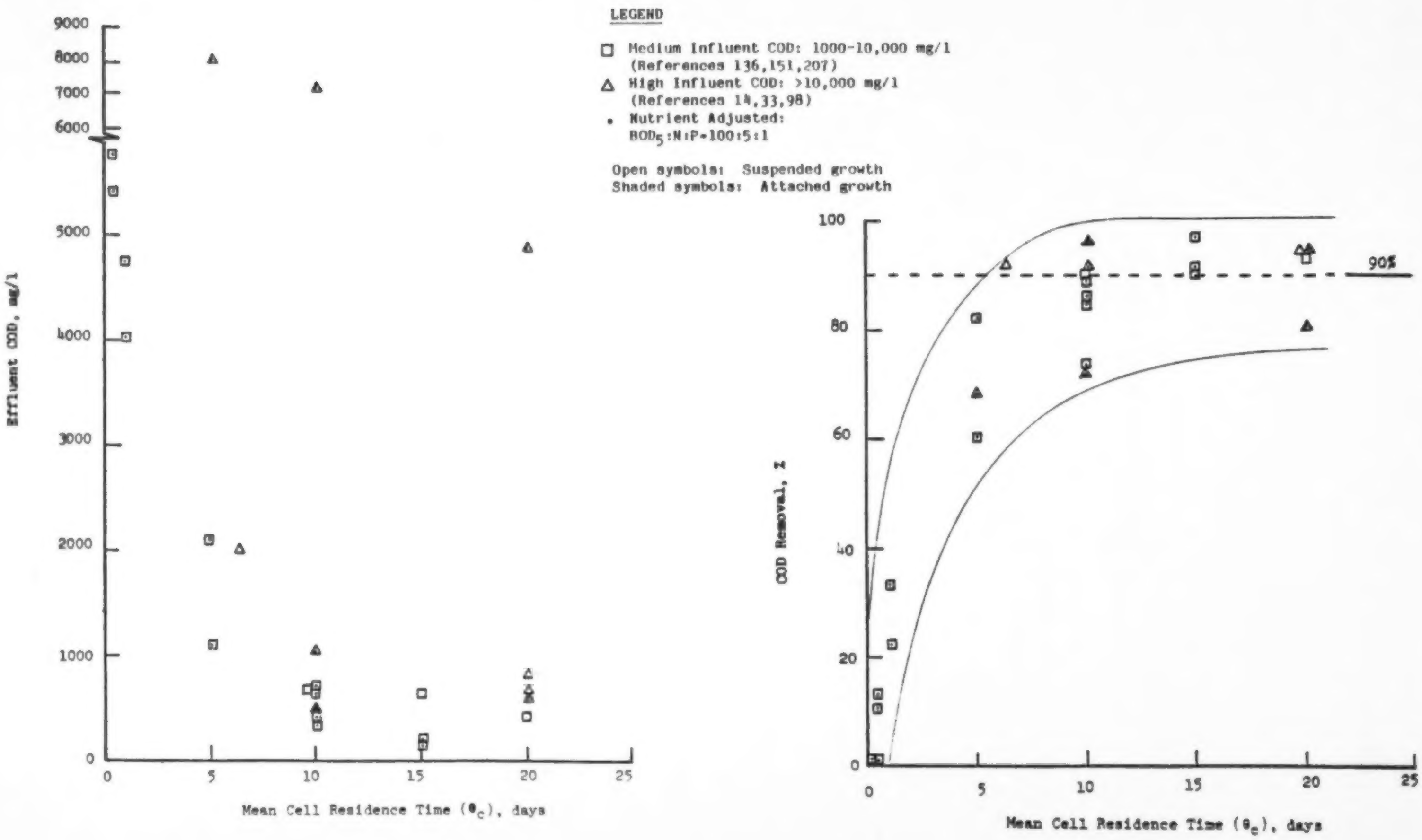

Figure 19. Relationship Between Mean Cell Residence Time and COD Removal for Bench-Scale Anaerobic Treatment Studies

also yielded similarly favorable results as summarized in Appendix Table A-9. In general, θ_c in excess of 10 days provided effluent BOD_5 and COD concentrations below 500 mg/l and 750 mg/l, respectively. These effluent levels were representative of 85 to 98% removal efficiencies.

The data were additionally segregated according to the previously used biodegradability ratios (BOD_5/COD and COD/TOC) as illustrated in Figure 20. Scrutiny of this figure confirmed the information already provided in Figure 19 in that greater than a 10 day θ_c allowed for better than 90% removal of influent COD.

Organic Loading Effects--

The relationships of effluent organics (BOD_5, COD) and organic removals to organic loading rate are illustrated for the data of Appendix Table A-10 in Figures 21 and 22. Process performance deteriorated rapidly beyond BOD_5 loadings of 1 kg/m^3·day or COD loadings of 2 kg/m^3·day. Treatment of high-strength leachates resulted in stronger effluents at high loading rates (1 kg BOD_5/m^3·day) than did medium-strength influents. However, higher percentage removals were recorded for the high-strength leachates at high loading rates than for the medium-strength leachates.

Gas production data are commonly used as an indication of process performance and are also of economic interest. Organic loadings were used as a basis for reviewing the quantitative significance of gas production data (Appendix Table A-11) during anaerobic treatment. The relationships of interest are illustrated in Figure 23 in terms of the volume of gas produced per kg of BOD_5 or COD destroyed at different organic loading rates. Considering the following redox stoichiometry at 35°C, the theoretical methane yield on a COD basis is 380 l/kg COD:

$$CH_4 + 2O_2 \longrightarrow CO_2 + 2H_2O$$

$$64 \text{ g } O_2/\text{mole } CH_4, \text{ or}$$

$$2.6 \text{ g } O_2/\text{liter } CH_4$$

Taking into consideration that gases produced will typically be on the order of 60 to 70% methane, the theoretical total gas yield for COD utilized is 550 to 650 l/kg COD. Furthermore, considering that the BOD_5/COD ratio for the anaerobic studies was typically 0.45 to 0.78, the theoretical gas yield on a BOD_5 basis would be on the order of 900 to 1000 l/kg BOD_5. These values are indicated by the dashed lines on Figure 23, which serve to illustrate the effects of increasing the loading rates beyond 5 kg COD/m^3·day or 2 kg BOD_5/m^3·day.

In comparison with the earlier plots of organic removal versus organic loading rate, the data of these figures seem to suggest that in the range of 2 to 5 kg COD/m^3·day, the gas production remains high, yet organics cannot be assimilated rapidly enough to avoid escaping into the effluent. Loading increases beyond 5 kg COD/m^3·day are apparently detrimental to the anaerobic methane-producing bacteria, as a result of substrate and/or chemical intermediate (volatile acids) induced inhibitions illustrated by decreasing gas production rates.

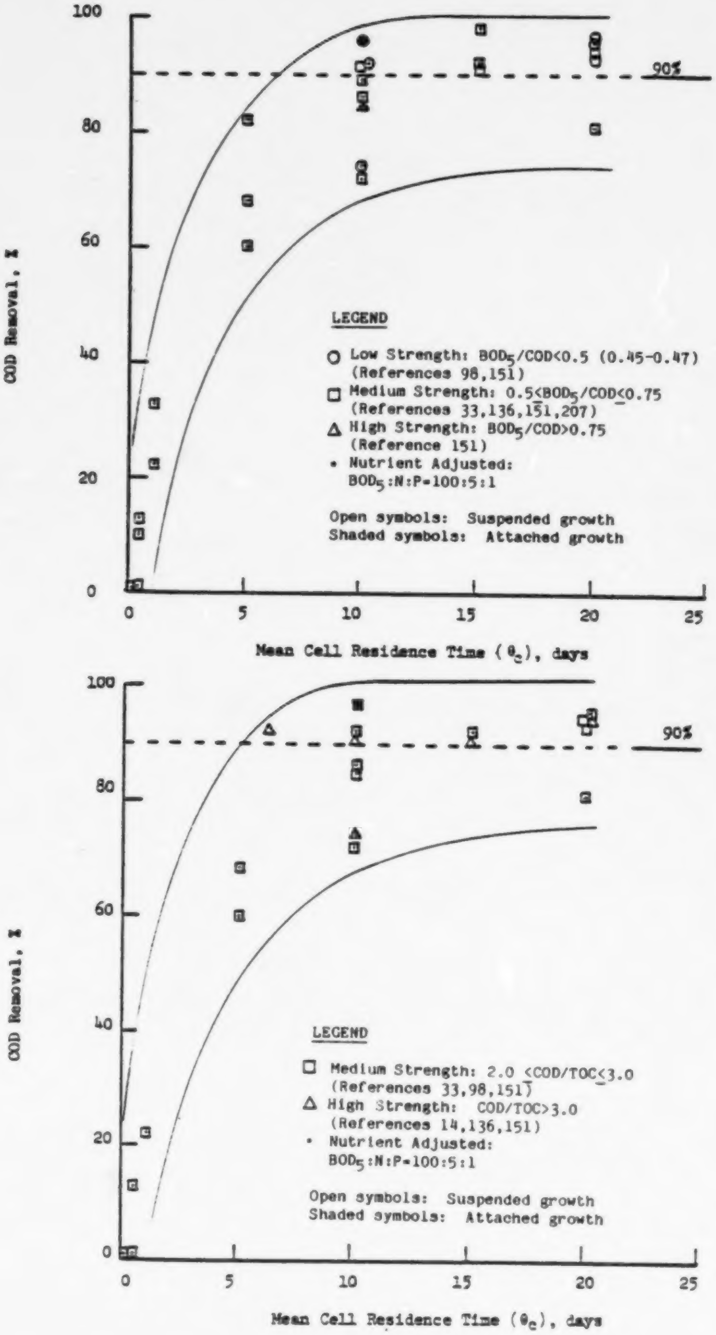

Figure 20. Illustration of COD Removal vs. θ_c for Anaerobic Treatment Data Segregated According to Biodegradability Ratios

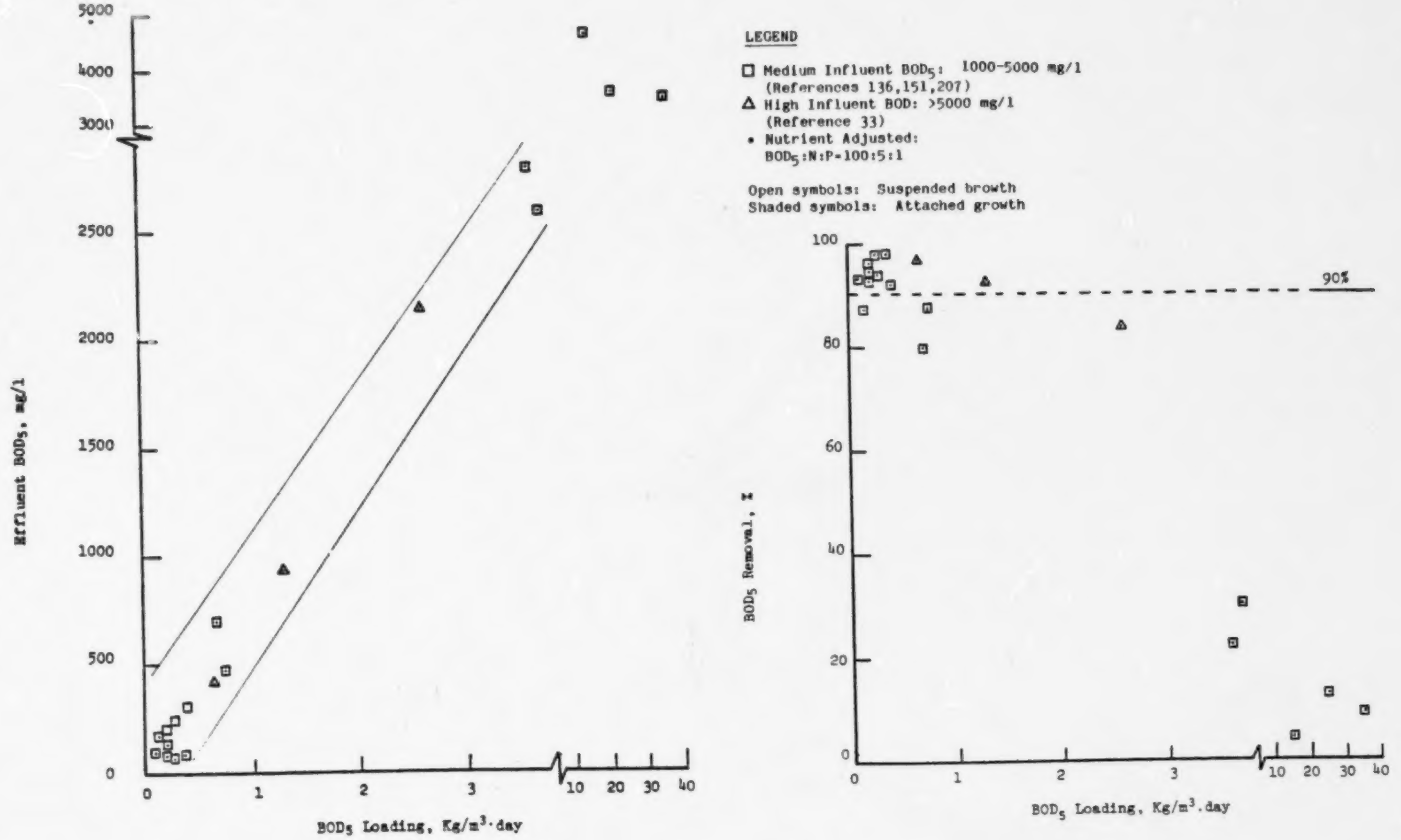

Figure 21. Relationship Between Organic Loading Rate and BOD5 Removal for Bench-Scale Anaerobic Treatment Studies

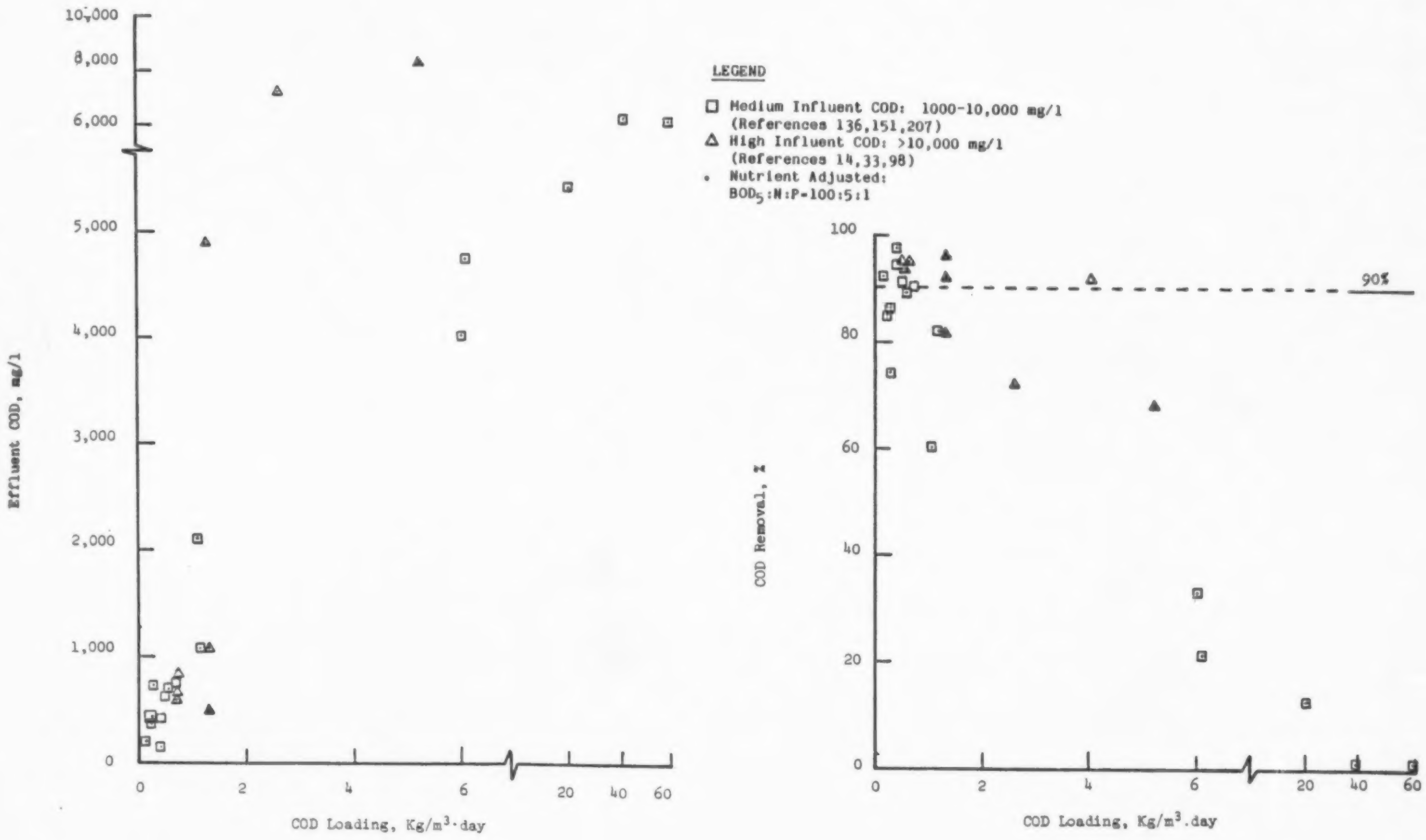

Figure 22. Relationship Between Organic Loading Rate and COD Removal for Bench-Scale Anaerobic Treatment Processes

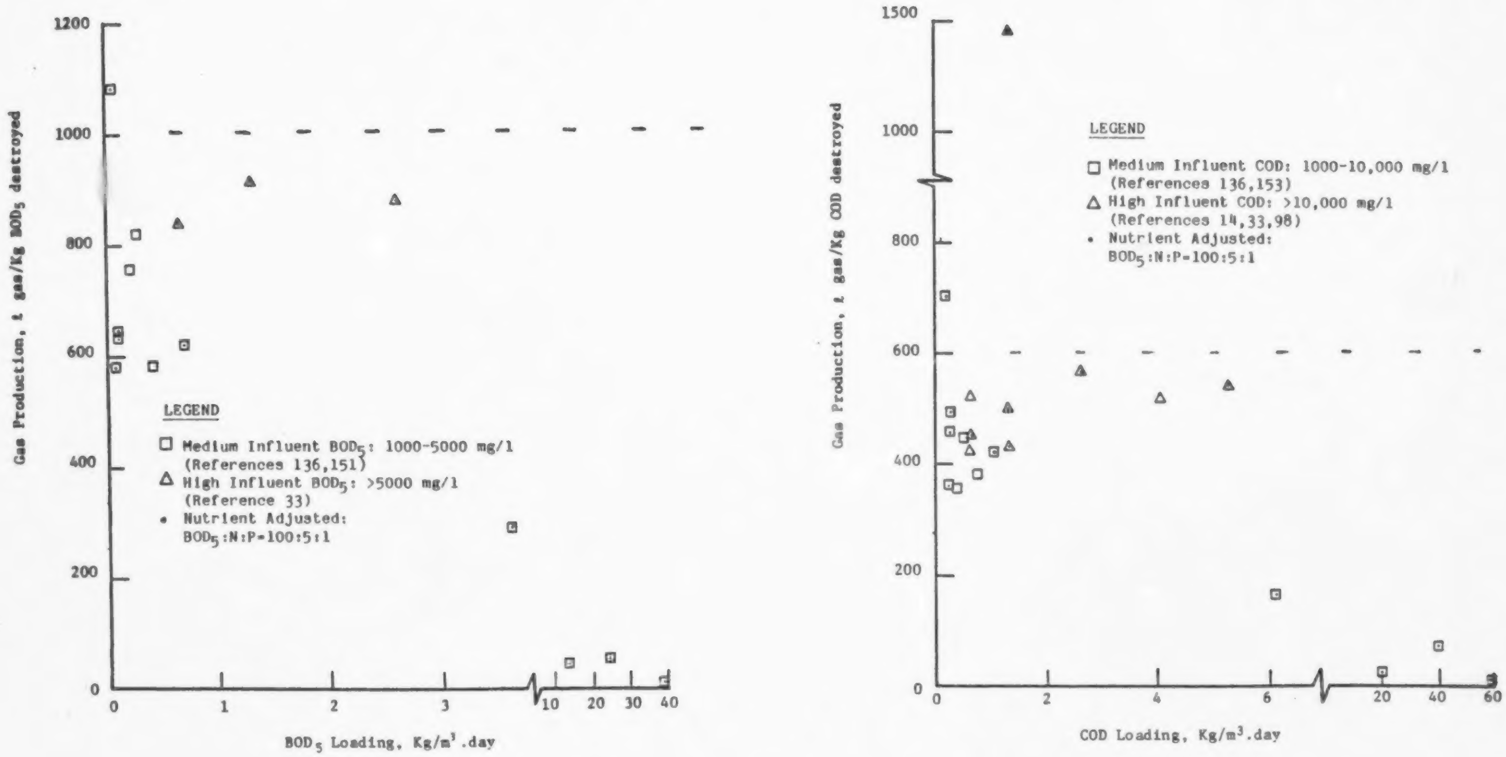

Figure 23. Relationship Between Gas Production and BOD5 and COD Loading Rates for Bench-Scale Anaerobic Treatment Studies

Temperature Effects--

From inspection of the data presented in Appendix Table A-9, successful anaerobic treatment was indicated at temperatures lower than the mesophilic range. Although studies on anaerobic treatment of leachate have been performed at temperatures ranging from 11° to 27°C, the most successful of these have been in the 23° to 27°C range as illustrated in Figure 24 for BOD_5 and COD removals.

The effects of temperature are further illustrated in Figure 25 using gas production. The figure clearly shows an increase in gas production at 33° to 37°C over that at 22° to 27°C. If sufficient retention time is provided (10-12 days), however, greater than 90% BOD_5 and COD removals can be realized with the lower temperature range. Moreover, the figures do not show a distinct difference between attached- or suspended-growth systems. (The numbers indicated next to the data points are their respective organic loading rates in kg $BOD_5/m^3 \cdot day$.)

Metals Removals--

A summary of metal removal data available from the literature for anaerobic treatment processes is presented in Table 17 and Appendix Tables A-12 and A-13. Except for iron and zinc, effluent heavy metal concentrations were generally on the order of 1 mg/l or less. As with the aerobic processes, zinc, iron, and chromium removals were above 90%. Copper, lead, cadmium, and nickel removals were on the order of 50 to 90%, although one study (Johansen, 1975) indicated no removal of cadmium and lead.

The alkali and alkaline earth metals were largely unaffected by anaerobic treatment processes with calcium being removed most efficiently, i.e., at 31%. Magnesium, potassium, and sodium removals were typically below 10% as indicated in Table 17 and Appendix Table A-13.

TABLE 17. SUMMARY OF HEAVY METAL AND ALKALI AND ALKALINE EARTH METAL REMOVAL DATA FOR BENCH-SCALE ANAEROBIC TREATMENT PROCESSES

Heavy Metals

	Cd	Cr	Cu	Fe	Pb	Mn	Ni	Zn
Influent Concentration Range, mg/l	0.03-0.1	0.22-1.7	0.03-5.6	245-810	0.12-1.4	6-18	0.19-1.2	5-15
Removal Range, %	0-99	0-90	38-88	80-99	0-84	69-92	10-86	80-99
Average Removal, %	14	73	60	95	13	81	68	96

Alkali and Alkaline Earth Metals

	Ca	Mg	K	Na
Influent Concentration Range, mg/l	315-1330	70-120	347-530	313-530
Removal Range, %	30-31	7-10	0-6	0-4
Average Removal, %	31	9	3	2

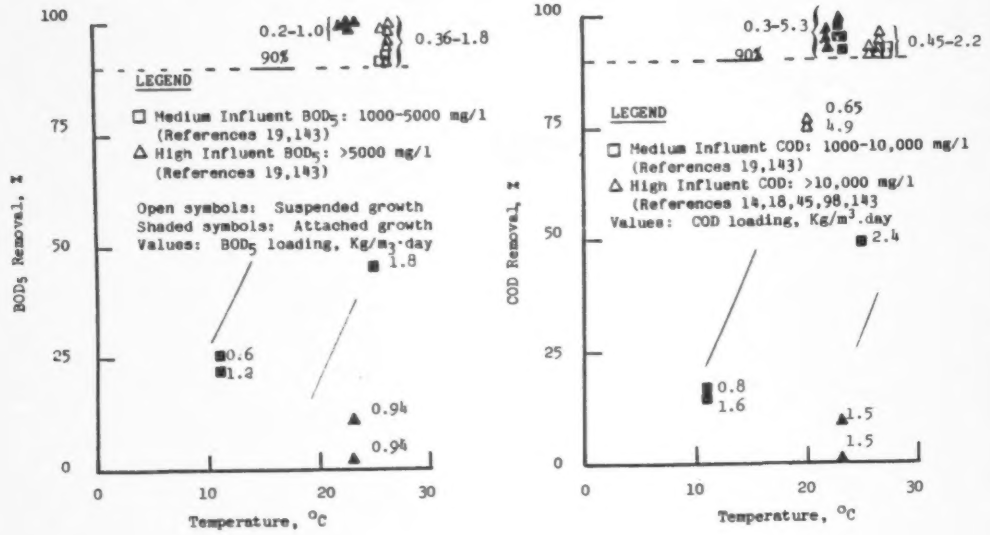

Figure 24. Relationship Between Temperature and Organics Removal for Bench-Scale Anaerobic Treatment of Leachate

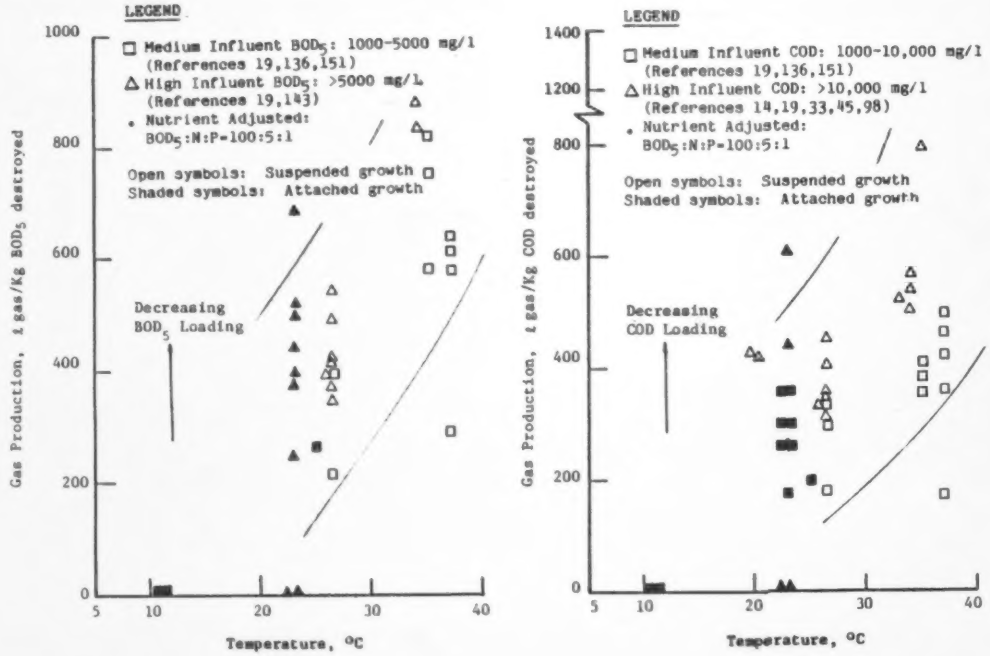

Figure 25. Relationship Between Temperature and Gas Production by Bench-Scale Anaerobic Treatment of Leachate

Anaerobic Treatment Kinetic Parameters--

The microbial dynamics of mixed reactor anaerobic processes can be described using a combination of Monod kinetics, cell yield, and mass balance concepts in the same fashion as for aerobic processes. Although limited data were available for the treatment of leachates, a summary of the parameters reported is presented in Table 18. These data compare fairly well with the kinetic data also presented in Table 18 for the conversion of acetic acid (a common leachate constituent). However, cell yields were somewhat higher for leachate treatment.

TABLE 18. SUMMARY OF MONOD KINETIC PARAMETERS FOR THE ANAEROBIC LEACHATE TREATMENT PROCESS

REFERENCE	INFLUENT CONCENTRATION, mg/l		Y, $\frac{\text{mgVSS}}{\text{mgBOD}_5 \text{ or COD}}$		b, day^{-1}		μ_{max}, $\frac{\text{mgBOD}_5 \text{ or COD}}{\text{mgVSS day}}$		K_s, mg/l	
	(BOD_5)	(COD)	(BOD_5)	(COD)	(BOD_5)	(COD)	(BOD_5)	(COD)	(BOD_5)	(COD)
33	13,000	26,000	0.1	-	0.006	-	5.9	-	4020	-
98	-	12,900	-	0.33	-	0.17	-	1.4	-	633
151, 205, 206	3,700	6,000	0.25**	0.14**	0.175	0.127	1.0	0.5	232	300
117*	-	1,600-2,100	-	0.04-0.07 (T=20°C)	-	0.03-0.05 (T=20°C)	-	0.31-0.38	-	13-165

*Nutrient adjusted fatty acid wastewater; BOD_5:N:P=100:5:1

**Based on dehydrogenase activity rather than VSS measurement as viable organism concentration

-Data not given

T = 34-37°C unless otherwise indicated

Anaerobic Process Sludge Characteristics--

Lower cell yields are generally exhibited by anaerobic processes when compared to aerobic processes, although this distinction is not exceedingly clear from a comparison of the yields reported in Tables 10 and 18. Limited sludge characterization performed on sludges resulting from the anaerobic treatment of leachate was available. However, some information on sludge solids volatility, percent solids, and metal content is summarized in Table 19. Scrutiny of this information indicates an average solids volatility on the order of 40% for solids contents of 2 to 7% (typically 4-5%). Iron and zinc were the most prevalent metals, existing in g/kg solids concentrations; calcium, chromium, copper, and lead were also occasionally found in high concentrations. No data on specific resistance, settling velocities, or sludge volume index (SVI) were located in the literature.

TABLE 19. SUMMARY OF SLUDGE CHARACTERISTICS FOR THE ANAEROBIC LEACHATE TREATMENT PROCESS

REFERENCE	REACTOR*	VOLATILE, %	SOLIDS, %	METAL CONTENT IN WET SLUDGE, $\frac{\text{mg metal}}{\text{kg SS}}$												COMMENTS
				Cd	Ca	Cr	Cu	Fe	Pb	Mg	Mn	Ni	K	Na	Zn	
14	PFR-SG	39	-	5.1	-	76	75	-	70	-	-	81	-	-	4300	T=20°C
	PFR-SG	-	-	-	-	-	-	$92x10^3$	-	-	-	140	-	-	2900	T=33°C
	PFR-SG	-	-	6	-	24	192	$43x10^3$	264	-	-	64	-	-	2700	T=30°C,Pilot Scale
	PFR-SG	-	-	16	-	64	192	$61x10^3$	264	-	-	112	-	-	3500	T=30°C,Pilot Scale
	PFR-SG	-	-	6	-	109	216	$77x10^3$	312	-	-	114	-	-	3150	T=30°C,Pilot Scale
33,217**	CSTR-SG	-	-	0.66	7600	4.6	4.0	70	19	390	321	7.2	625	750	2330	T=34°C
	CSTR-SG	-	-	<0.7	4100	1.1	1.8	5230	3.6	317	114	4.1	1290	588	1100	T=34°C
143	PFR-AG	36	6.9	40	-	300	900	300	100	-	-	-	-	-	2500	T=23°C
	PFR-AG	37	5.4	40	-	300	600	300	100	-	-	-	-	-	5000	T=23°C
	PFR-AG	36	7.2	40	-	300	1100	300	100	-	-	-	-	-	4100	T=23°C
	PFR-AG	51	0.95	700	-	5800	4100	5000	2100	-	-	-	-	-	$16x10^3$	T=23°C
45,46,73	CSTR-AG	38	3.9	-	-	-	-	-	-	-	-	-	-	-	-	T=23°C ρ=1.026 g/cm^3
Average	-	40	5.0	100	-	900	900	$35x10^3$	400	-	-	100	-	-	4900	

- Data not given
* PFR-SG = Plug flow reactor suspended growth
CSTR-SG = Continuously-stirred tank reactor suspended growth
PFR-AG = Plug flow reactor-attached growth
** Dry sludge (centrifuged)

In situ Anaerobic Leachate Recycle Treatment

The collection and recycle of leachate back onto or into a landfill represents an in situ method of leachate treatment as opposed to the other biological processes previously reviewed. The treatment mode involved in this approach is primarily anaerobic, although aerobic conditions at the beginning and formation of humic substances during the final phases of a landfill's "life" may be important with regard to organic conversion and the possible re-mobilization of heavy metals, respectively. Moreover, the practice of recycling leachate serves to improve the homogeneity of the biochemical environment needed for anaerobic waste degradation, and may, thereby, effectively shorten the time normally required for waste "stabilization" by as much as 80 to 90% (Pohland, 1975, 1980). Current evidence also suggests that the costs of leachate recycle treatment may be as much as 25% of the costs of corresponding separate treatment (Pohland, 1979).

Pilot-Scale Leachate Recycle--

A number of pilot-scale investigations on the application of leachate recycle have been performed. The test cells utilized and research objectives associated with these studies are presented in Table 20. Operating variables such as moisture content, pH adjustment, nutrients, microbial seed, and the use of recycle have been reported as indicated in Table 21. Of these variables, the use of recycle and buffers have emerged as most important in accelerating the onset of anaerobic waste degradation and in maximizing the rate, consistency and quality of gases produced.

While recycle and buffer addition served to significantly shorten the stabilization period, the effluent concentrations ultimately obtained by comparative cells utilizing nutrients and/or microbial seedings, but without recycle, were very similar. The effluents ultimately obtained were also very similar in character to those obtained from anaerobic treatment processes, e.g., BOD, ~100 mg/l; COD, ~300 to 500 mg/l; TKN, ~100 to 300 mg/l; and, Fe, ~540 mg/l.

In general, recycling of leachate promoted the development of in situ biological, physical and chemical mechanisms responsible for waste stabilization and/or leachate treatment. Biological assimilation of the organic substrate by anaerobic microbial processes resulted in residual BOD_5 and COD concentrations of 30 to 500 mg/l and 70 to 800 mg/l, respectively (Table 21). Moreover, as microbial degradation progressed, the nature of organic substrates in leachates became more refractory, as indicated by the low BOD_5/COD and COD/TOC ratios of 0.15 to 0.4 and 0.9 to 1.9, respectively. TKN removal by leachate recycle effective with residual concentration of ~50 to 100 mg/l being typically achieved in long-term leachate recycle studies. The pH of recycled leachate eventually increased to a range of 6.5 to 7.0 as a result of the volatile fatty acids assimilation during the biodegradation process.

The removal of heavy metals, represented by Fe and Zn in Table 21, was also effective. Residual concentrations of 40 mg/l Fe and 4.0 mg/l Zn were commonly reported for the recycled leachate. The efficient removal of heavy metals was attributed to chemical complexation by inorganic and organic ligands which were found to be abundant in leachate and were able to form metal-ligand precipitates. Sulfides were also determined to be a significant

TABLE 20. PILOT-SCALE RESEARCH PERFORMED ON LEACHATE TREATMENT BY LEACHATE RECYCLE

REFERENCE	TEST UNIT	PROCESS DESCRIPTION	RESEARCH OBJECTIVE(S)	LEACHATE SOURCE
16	3m high column lysimeter	Six columns with different waste mixtures, organic and inorganic wastes.	Determine treatability of leachate from pulp and paper mill wastes through recycle; evaluate organic and metal removal and gas production.	Pulp and paper mill waste.
82-84,167	15m square by 3m high test cell	Five test cells; control (no recycle), high initial moisture content, continuous flow through of water. leachate recycle, and biological sludge seeding with high initial moisture.	Determine feasibility of leachate recycle for refuse stabilization; effect of once through moisture; effect of biological sludge seeding.	Municipal solid waste.
8,22,174,177, 204-207	3m high column lysimeter	Four columns: control (no recycle), recycle, recycle with pH control at neutral pH, and recycle with pH control at neutral pH with biological sludge seeding.	Effect of pH control and biological sludge seeding on organic stabilization of waste.	Municipal solid waste.
208,211	5m square by 3m high test cell	Two cells; one sealed to prevent evaporation, other open to atmosphere to allow for evaporation. Both received equivalent amount of water from rainfall.	Effect of evaporation on refuse stabilization by recycle. Evaluate organic removal and gas production.	Municipal solid waste.
210,212	3m high column lysimeter	Four columns: control (municipal solid waste only) and three with different quantities of plating wastes mixed with municipal solid waste.	Determine removal mechanisms of metal ions by studying chemical activity and chemical complexation.	Municipal solid waste with metal plating wastes.

TABLE 20 (Continued)

280	1.8m high, 0.9m dia. test cells	Sixteen cells; combination of recycle, buffers, nutrient additions to leachates.	Determine the effects of moisture, recycle, pH, and nutrients on gas production and leachate quality.	Municipal solid waste.
57	1.5m by 1.5m square test cells	Four cells; recycle of leachate plus annual rainfall, no recycle, recycle of half the annual rainfall, and, presaturation followed by recycle of half the annual rainfall.	Determine the effects of moisture content and leachate recycle on gas production and leachate quality.	Municipal solid waste.
57	$600m^3$ test fields 15mx10mx4m	Three fields filled with compacted wastes; one sealed against evaporation and recycled, one with recycle and no seal, one without recycle.	Determine the effects of moisture content and leachate recycle on gas production and leachate quality.	Municipal solid waste.
221	1.6m deep by $5m^2$ area test cells	Three test cells; simulated annual rainfall applied to each; one with leachate recycle, one with recycle of aerated leachate, one without recycle.	Determine effects of leachate recycle on gas production and leachate quality; effects of leachate aeration and phosphorous addition on *in situ* biodegradation.	Municipal solid waste.

TABLE 21. SUMMARY OF TEST VARIABLES, LEACHATE CHARACTER, AND GAS RESULTS FOR THE PILOT-SCALE LEACHATE RECYCLE STUDIES

REFERENCE	TEST VARIABLES*								RECYCLE FREQUENCY	TEST PERIOD, days	LEACHATE CHARACTER AT END OF RECYCLE PERIOD**								^{3}GAS YIELD, m^3/1000 kg dry	CH_4,%
	C	F	M	N	pH	R	S	Sh			BOD_5	COD	BOD_5/COD	COD/TOC	TKN	Fe	Zn	pH		
82-84,	X									1440	40,000	50,000	0.8	-	500	1200	50	5.5	-	1
167			X							1440	30,000	50,000	0.6	-	1000	300	70	5.5	-	20
		X								1440	3000	5000	0.6	-	100	200	1.0	6.2	-	65
						X			Daily	1440	400	1500	0.3	-	200	50	1.0	6.5	-	65
			X				X				30,000	50,000	0.6	-	1000	800	80	5.5	-	20
8,22,174,	X									1063	2000	3500	0.6	2.0	11	450	15	5.8	-	-
177,204-207						X			-	1063	30	70	0.4	1.8	2.0	4	-	6.8	-	-
					X	X			-	747	35	240	0.15	1.0	8.5	3	0.2	7.0	-	65
					X	X	X		-	747	40	170	0.2	1.3	1.4	9	0.4	7.0	-	65
208,211	(open cell)					X			Weekly then daily	492	90	350	0.3	1.0	20	39	0.5	6.7	-	-
	(sealed cell)					X				492	70	300	0.2	0.9	20	29	0.2	6.6	7.1	55
266,268	X									514	8000	10,000	0.8	2.5	-	-	-	-	-	-
						X			Daily	368	120	150	0.8	1.7	50	7	1.0	4.3	-	-
					X	X		X	Daily	514	350	500	0.7	1.9	20	3.5	0.2	7.0	-	-
					X	X			Daily	514	200	350	0.6	3.0	36	7	0.2	7.0	-	-
				X	X	X			Daily	514	200	350	0.6	3.0	330	6	0.7	7.0	-	-
						X			Daily	400	36,000	61,290	0.6	-	-	-	-	6.3	-	55
57	X									400	38,300	62,690	0.6	-	-	-	-	6.5	-	45
						X			Daily	400	39,700	66,310	0.5	-	-	-	-	6.3	-	45
		X				X			Daily	400	35,000	58,330	0.6	-	-	-	-	6.3	-	50
280		X								720	-	12,000	-	2.5	250	-	-	6.2	14	50
		X			X					720	-	16,00	-	2.4	170	-	-	5.9	14	65
	X								Daily	720	-	36,000	-	2.1	825	-	-	6.3	15	65
	X				X				Daily	720	-	26,000	-	2.2	875	-	-	6.8	17	70
221		X								900	-	-	-	-	-	-	-	-	-	-
						X			Daily	900	33	600		3.0	136	42	0.14	7.1	-	-
					X*	X			Daily	900	20	618		3.0	50	50	0.19	7.0	-	-

*C = Control. No recycle and no water addition.
F = Flow through of water without recycle.
M = Moisture added initially.
N = Nutrient addition.
pH = Adjustment to neutral pH.
R = Recycle of leachate.
S = Sludge seed added.
Sh = Shredded solid waste.
** All expressed in mg/l except BOD_5/COD, COD/TOC, and pH.
- Data not given

factor in precipitation of heavy metals, with the possible exception of cadmium which was not as readily precipitated (Pohland, et al., 1981). Leachate recycle resulted in a gas yield of 7.1 m^3/1000 kg dry waste (Pohland, 1980) with a gas composition of 55 to 65% CH_4 and 35 to 45% CO_2 as also reported in two other studies included in Table 21.

Full-Scale Leachate Recycle--

As yet, no full-scale testing of leachate recycle as an in situ treatment option in the United States has been reported in the available literature. One full-scale study has been performed in England, and several full-scale landfills have been provided with leachate recycle in Germany.

A demonstration project has been conducted which may be considered near full-scale at Mountain View, CA (Pacey, 1983). This study was conceived to verify pilot-scale observations regarding the benefits of adequate moisture content, pH buffering and nutrient availability through controlled moisture applications and/or leachate recycle. Six field cells were constructed to evaluate these effects, each having an average volume of 10,500 m^3 and refuse mass of 4825 metric tons. Each of these cells was operated using different combinations of water content, seed sludge, nutrients, and buffer. Only one of the six cells was operated using leachate recirculation. Unfortunately, the initial moisture application to this latter cell was somewhat drastic and was followed by an infrequent and sporadic leachate recycle schedule which tended to obscure the benefits of leachate recycle (Van Heuit, 1983; Pacey, 1983). Although still somewhat preliminary, the results of this study have illustrated the benefits of pH and moisture control, i.e., cells to which moisture and buffer were applied have produced significantly higher quantities of gas than the control cells. Despite sporadic recirculation, the recycle cell has produced the highest quantities of gas to date. Routine leachate quality was not monitored, therefore, definitive conclusions regarding stabilization patterns from this study were difficult.

A 2.5-ha landfill in England has been lined with a heavy polyethylene membrane and filled to a depth of 3 to 4 m with refuse having a density of 800 to 1000 kg/m^3 (Robinson, et al., 1982). Leachate has been sprayed on the top of the refuse using a sprinkler system. Preliminary results have indicated that the COD of recirculated leachate is diminishing at a significantly higher rate (40% reduction in the first 20 months of operation) than in a non-recirculated control area. Unfortunately, gas production data were not available since the landfill was not covered.

Some information is available for several full-scale landfills in Germany where leachate recycle is being used as summarized in Table 22 (Cord-Landwher, et al., 1982). A two-stage approach was initiated at one of these landfills wherein leachate was removed from a newer landfill section to be recirculated in an older stablilized section. The 'acid-stage' (new field) had a surface area of 0.6 ha and a refuse depth of 4 m; the 'methane-stage' (old field) had an area of 0.57 ha and a similar depth. Eight months of operating data for this system have been presented and are summarized in terms of leachate BOD_5 and COD in Table 23. Results indicated that the two-stage approach may be used to obtain consistent quantities of methane from a full-scale landfill at a minimum cost, since the gas collection and leachate recirculation systems would not require as frequent or extensive modifications as in the case where the total landfill would be filled to capacity. In the staged approach,

TABLE 22. SUMMARY OF AVAILABLE INFORMATION CONCERNING THE APPLICATION OF LEACHATE RECYCLE AT FULL-SCALE LANDFILLS IN GERMANY

Landfill	Stapelfeld	Flechum	Dorpen	Venneberg (Old)	Venneberg (New)	Hattorf	Blankenhagen	Nauroth	Reinstetten	Kupferzell Betterspot
Surface Area, ha	8	2.46	5.5	6.0		9.4	18.0	7	5.4	3
Area served, km^2	6	8	9	3	9	12	12	30	15	12
Population served	70,000	18,000	80,000	82,100		115,340	160,000	122	112,620	85,000
Refuse Received										
kiloton/yr	-	-	-	-		-	-	-	62,000	51,000
m3/yr	-	17,000	70,000	65,000		100,000	180,000	120,000	-	-
Leachate produced										
m3/yr		1,000	570	-		1,800	7,630	-	-	-
m3/ha-yr	-	-	-	1,400	1,590	-	-	-	1,574	-
Recirculation method										
sprinklers	x	-	-	x		-	-	x	x	x
troughs	-	x	x	-		x	-	-	-	-
others	-	-	-	-		-	x	-	-	x
Leachate quality										
BOD$_5$,mg/L	820	-	-	100	20,000	390	1,400	200	-	140
COD,mg/L	1,680	-	-	1,200	35,000	930	2,900	-	-	48,000
pH	7.4			7.8	6.1	6.95	7.65	7.4	-	8.1
Start of landfill	1973	1975	1979	1976		1977	1962	1973	1975	1980
Start of recycle	1981	-	-	-		-	-	1975	1982	1980
Annual precipitation,mm	750	810	700	790		-	650	1100	800	650

After Cord-Landwher, (1982)

leachates collected from sections of the landfill which have not been equipped with recirculation or gas collection appurtenances can be stabilized by the methanogenic bacteria operative in the older sections. Moreover, this method may significantly improve the overall yield of available methane from a landfill while affording a lower capital investment.

TABLE 23. ORGANIC CHARACTERISTICS OF LEACHATES REMOVED FROM A FULL-SCALE TWO-STAGE RECIRCULATION PROCESS IN GERMANY

Date of Sample (1982)	Old Field (Methane-Stage)		New Field (Acid-Stage)	
	BOD_5, mg/l	COD, mg/l	BOD_5, mg/l	COD, mg/l
2/17	60	1473	--	--
3/3	64	1278	1,310	5,303
4/15	59	1370	5,320	10,390
5/12	--	--	2,660	5,559
6/3	63	1561	6,000	16,725
7/7	67	1409	--	--
7/22	--	--	6,340	11,200
8/3	60	1273	11,970	19,300

After Cord-Landwher, *et al.*, 1982

PHYSICAL/CHEMICAL TREATMENT OF LEACHATES

A number of physical/chemical processes have been investigated for their respective leachate treatment capabilities. Much of the emphasis has been on bench-scale, although several processes have also been evaluated on full-scale. Bench-scale investigations have included the application of chemical oxidation, precipitation, coagulation, ionizing radiation, ion exchange, adsorption and reverse osmosis. Full-scale investigations have been performed on chemical precipitation/coagulation, ammonia stripping, and activated carbon adsorption.

Bench-Scale Physical/Chemical Leachate Treatment Processes

A listing of bench-scale research activities on the reported physical/chemical leachate processes is provided in Table 24 together with descriptions of the processes used and the objectives of each study.

Coagulation and Precipitation--

Organics Removal--Coagulation and precipitation have been the most extensively studied physical/chemical treatment methods for the removal of organics and metals. Alum, ferric chloride, lime, and polymers have been used as coagulants as summarized in Appendix Table B-1. As shown in Figure 26, none of the coagulants tested have been successful in removing more than 30% of the influent COD from either raw or biologically treated leachates.

TABLE 24. BENCH-SCALE RESEARCH PERFORMED ON LEACHATE TREATMENT BY PHYSICAL/CHEMICAL PROCESSES.

Reference	Process	Process Description	Research Objective(s)	Leachate Source*
17,18,176	Oxidation Disinfection Precipitation	Oxidation and disinfection by diffusing ozone in batch test; precipitation by standard batch jar test apparatus simulating coagulation, flocculation, and settling.	Effect of ozone on oxidation of organics and disinfection; effect of lime addition on organic and metal removal	Lysimeter
19,20	Coagulation Oxidation Precipitation	Standard batch jar test apparatus to simulate coagulation, precipitation and settling.	Determine optimum dosage for organic, iron, and color removal using $FeCl_3$ and alum as coagulants, Cl_2 and $KMnO_4$ as oxidants, lime and Na_2S_2 as precipitants.	Landfill
32,56,170	Adsorption Coagulation Precipitation	Continuous flow adsorption column; standard batch jar tests to simulate coagulation, precipitation and settling.	Effect of peat adsorption for removal of organic matter and metals; determine optimum dosage or heavy metal removal using $FeCl_3$ as coagulant and lime and NaOH as precipitants.	Landfill
44-46,70,73	Adsorption Ion Exchange Oxidation Precipitation Reverse Osmosis	Continuous flow column test and batch tests for AC adsorption; complete-mix, batch reactor for O_3 oxidation; standard batch jar test for lime precipitation	Determine optimum process for removal of organic matter using AC, anion exchange resin, ozone, lime, and reverse osmosis for raw leachate and biologically treated effluents.	Landfill and Lysimeter
54,97	Adsorption Coagulation Oxidation Precipitation	Continuous flow column test for AC adsorption; standard batch jar test apparatus for coagulation, oxidation precipitation.	Evaluate effect of color removal for effluent polishing using AC adsorption and NaOCl oxidation; determining optimum dosage for organic removal using $FeCl_3$, $FeSO_4$, alum, and polymer as coagulants and lime and NaOH as precipitants for raw leachate.	Landfill
118,119,176	Precipitation	Standard batch jar test.	Determine optimum lime dosage for organic removal from biologically treated effluent.	Lysimeter
133	Adsorption Coagulation	Continuous flow column test for AC adsorption; standard batch jar test for alum coagulation.	Determine alum dosage and AC effectiveness for organic and heavy metal removal for conceptual design of full scale treatment plant.	Landfill
134	Adsorption Coagulation Oxidation Precipitation	Batch test and continuous-flow column test for AC adsorption; all other tests performed on a batch basis.	Effect on organic, iron, and removal by AC adsorption, alum and $FeCl_3$ as coagulants; $Ca(OCl)_2$. Cl_2, $KMnO_4$ and O_3 as oxidants; and lime and Na_2S as precipitants.	Landfill and waste pile
229	Adsorption Oxidation	Batch AC adsorption test; Batch ozone oxidation test.	Effect of AC adsorption and and ozonation on removal of organics, phenol, NH_3, and toxic organics.	Lysimeter
242,243	Adsorption Ion Exchange	Continuous upflow filters for both AC and glauconitic greensand.	Effect of AC adsorption and greensand for metal removal. Evaluate effect of process sequence between adsorption and greensand ion exchange.	Landfill
265	Coagulation Precipitation	Standard batch jar test apparatus.	Determine optimum dosage of alum coagulant and lime precipitant for organic, color, and metal removal.	

TABLE 24 (Continued)

Reference	Process	Process Description	Research Objective(s)	Leachate Source*
183,205-207	Adsorption Ion Exchange	Batch study for AC adsorption tion; Batch study for ion exchange using cation resin.	Determine treatability of aerobically treated leachate effluent using cation resin, mixed resin and PAC.	Landfill and Lysimeter
151	Adsorption Coagulation Oxidation Precipitation	Batch test for AC adsorption; Standard batch jar test for alum and lime, $FeSO_4$ and lime, and NaOCl dosages.	Determine adsorption capacity for AC on raw leachate; Determine optimum dosage for alum, lime, $FeSO_4$, and NaOCl for organic and metal removal.	Landfill
143	Adsorption Coagulation Precipitation	Batch test for AC adsorption using jar test apparatus.	Determine adsorption capacity of AC for chemically treated leachate and biologically and chemically treated leacahte; Determine optimum dose of alum, $FeCl_3$, and lime and optimum pH for organic, Fe and Zn removal for raw and biologically treated leachate.	Landfill
28	Adsorption Coagulation Precipitation	Continuous flow column test and batch tests for AC adsorption; standard jar test to simulate coagulation, precipitation, and settling.	Compare effectiveness of three activated carbons for polishing chemically and biologically treated leachate; determine optimum dosage of alum, lime, and ferric chloride for organic and iron removal.	Lysimeter
158	Adsorption Coagulation	Continuous flow column test for AC adsorption; standard jar test for ferric chloride coagulation.	Determine AC effectiveness and and optimum dosage of $FeCl_3$ for organic removal.	Landfill
285	Ionizing Radiation	Radioactive isotope of cobalt used as gamma ray source for ionizing organic substrate.	Effect of pH, aeration rate, and dose rate on organic removal; determine molecular weight distribution of ionized organics.	Landfill
238	Coagulation Precipitation	Standard batch jar test to to simulate coagulation, precipitation, and settling.	Determine optimum dosage of alum and $FeCl_3$ as coagulants and lime and NaOH as precipitants for organic and heavy metal removal.	Landfill
215,216	Disinfection	Batch reactor using NaOCl as as disinfectant.	Effect of NaOCl dosage and contact time on bacterial and viral inactivation.	Landfill
96	Coagulation	Standard Batch jar test to simulate coagulation and settling.	Determine optimum dosage for organic and iron removal using alum as coagulant.	Lysimeter
239	Coagulation	Standard Batch jar test to simulate coagulation and settling.	Determine effects of lime dosage on color, turbidity, and organics removal.	Industri Landfill

The effects of coagulant dosage and pH are illustrated in Figure 26. Alum has been demonstrated as the most successful coagulant in dosages of 50 to 100 mg/l and at a pH near 8.2, achieving up to 25% COD removal. Ferric chloride and polymer were determined to be somewhat less effective at similar and greater dosages. For these coagulants, COD removals were typically on the order of 10% in the pH range of 6 to 9 and at coagulant dosages up to 1000 mg/l.

Results of chemical precipitation using lime, sodium hydroxide, and sodium sulfide are summarized in Appendix Table B-2 and illustrated in Figure 25. The data presented in Figure 27 indicate that chemical precipitation processes were equally unsuccessful in removing COD. Lime dosages of 1000 mg/l resulted in only 25% COD removals from raw leachate at pH 7. Similar dosages for biologically treated leachates yielded 35% COD removals. Although the use of sodium sulfide and sodium hydroxide received only limited study, results indicated that less than 10% COD removal was possible at chemical dosages upwards of 2000 mg/l.

Metals Removal--Alum, ferric chloride, and lime have been investigated for their respective metal removal potentials as indicated in Appendix Table B-3 for the heavy metals and in Appendix Table B-4 for the alkaline earth metals. As summarized in Table 25, iron and zinc were removed best with 90% or greater removals being generally achieved. Alum and ferric chloride at dosages of less than 100 mg/l have provided successful removals, whereas much higher dosages of lime (500 mg/l) were required to achieve similar results. Data for only one analysis with sodium sulfide indicated 99% iron removal at a 1000 mg/l dosage.

TABLE 25. SUMMARY OF HEAVY METAL AND ALKALI AND ALKALINE EARTH METAL REMOVAL DATA FOR BENCH-SCALE CHEMICAL ADDITION PROCESSES

Heavy Metals

	Cd	Cr	Cu	Fe	Mn	Pb	Ni	Zn
Concentration Range, mg/l	-	0.08-0.064	0.035-0.56	317-1000	0.7-25	0.10	73	0.4-30
Removal Range, %	-	30-53	21-96	0-99	28-99	20	4	0-99
Average Removal, %	-	40	40	84	66	20	4	86

Alkali and Alkaline Earth Metals

	Ca	Mg	K	Na
Concentration Range, mg/l	178	100-160	156-350	188
Removal Range, %	0-6	0-60	8-27	43
Average Removal, %	2	15	19	42

- Data not given

Lime has been shown capable of removing manganese, potassium, and sodium, although the dosages of lime required generally greatly exceeded the quantities of metals removed as shown in Appendix Table B-4. Alum, ferric chloride and ferrous sulfate have received only limited study for the removal

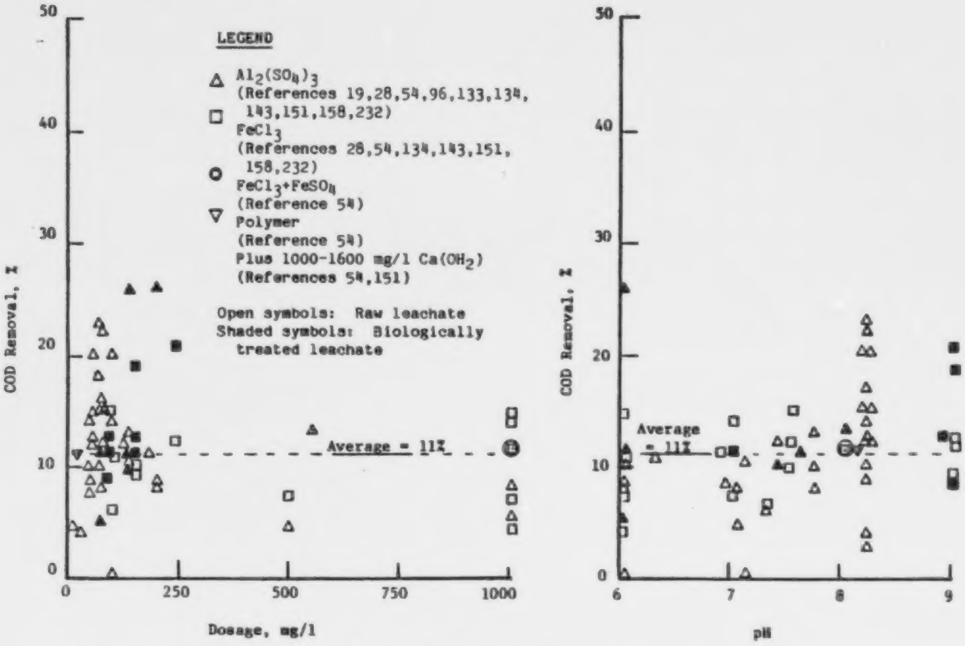

Figure 26. Relationships Between pH and Dosages of Various Chemical Coagulants and Corresponding COD Removals

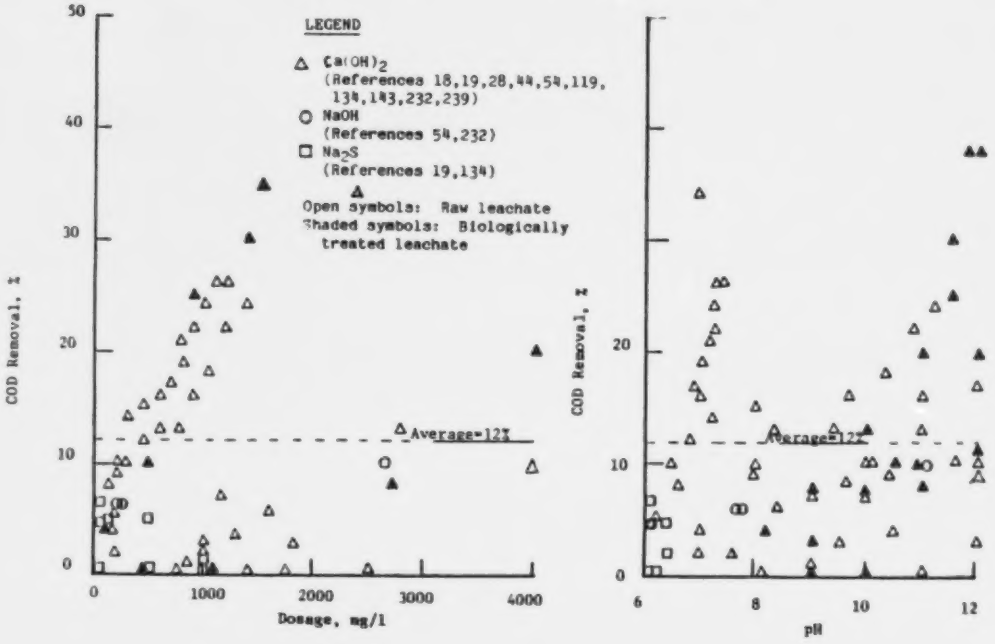

Figure 27. Relationships Between pH and Dosages of Various Chemical Precipitants and Corresponding COD Removals

of alkaline earth metals. From the limited data given in Appendix Table B-4, large doses of these chemicals were relatively unsuccessful in removing the alkaline earth metals indicated.

Chemical Oxidation--

Chemical oxidation of leachate organics has been investigated using chlorine, calcium hypochlorite, sodium hypochlorite, potassium permanganate, and ozone. In general, chemical oxidation processes have been slightly more successful than the chemical coagulation and precipitation processes for COD removal, but removal efficiencies have been too low to be considered practical. As shown in Figure 28, 10 to 30% COD removal was typically achieved with dosages of 2000 mg/l of NaOCl, Ca(OCl), and Cl_2. Effects of ozone were similar at lower dosages, but retention times of 3 to 4 hours were required as summarized in Appendix Table B-5. Hypochlorites were somewhat superior to the other oxidants studied with regard to COD removal. However, the hypochlorite dosages required were exceedingly high.

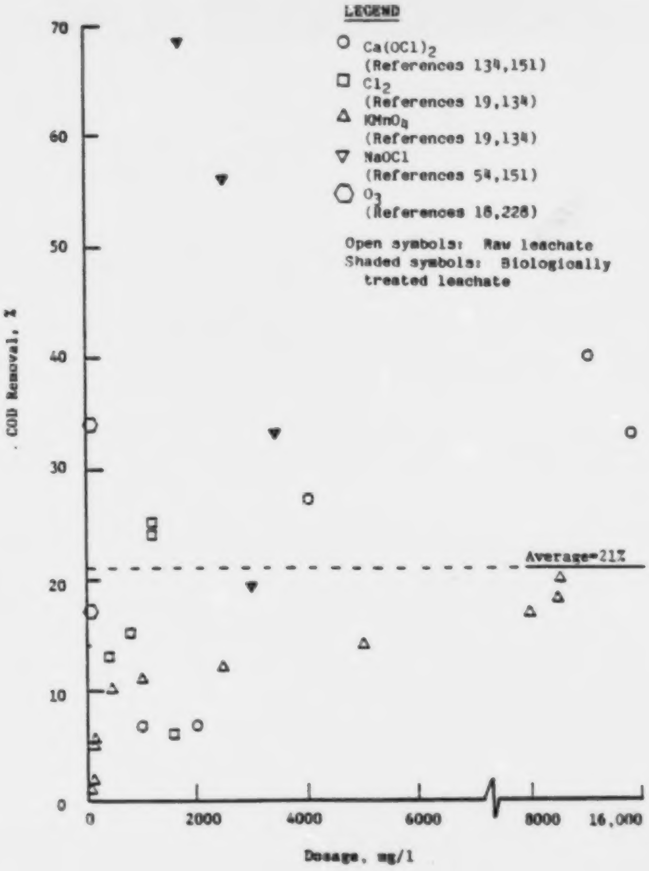

Figure 28. Relationship Between Chemical Oxidant Dosage and COD Removal for Bench-Scale Chemical Oxidation Studies

Only one chemical oxidant, NaOCl, has been tested for the treatment of biologically treated leachates. The application of this oxidant was successful in removing 20 to 70% of the residual organics (as COD) from a biological process effluent. Although limited data were available, the best removal (69%) was achieved at the lowest dosage (1600 mg/l) and better results were also observed at pH 8.9 than pH 9.5 or above. Presumably, chemical oxidation would be more logically used for this application (treatment of biological process effluent), since the stronger oxidants would tend to convert the refractory organics remaining after biological treatment.

Metal removal by chemical oxidation processes was studied by only a few researchers. As shown in Appendix Table B-3, ozone treatment was successful in removing 82 to 99% of iron, copper, and zinc. However, nickel was not removed. The application of chlorine compounds and chlorine were also very successful in removing iron, achieving 99% or better removal with dosages of 800 to 1000 mg/l.

Chemical Disinfection--

Ozone and sodium hypochlorite have been applied to raw and biologically treated leachates to evaluate their capabilities for disinfection. As shown in Table 26, ozone at 100 mg/l decreased the bacterial density of raw leachate to 30 CFU/ml, as determined by the Standard Plate Count Technique. This dosage is two orders of magnitude higher than typically reported for domestic wastewater disinfection (Venosa, 1972), since the high level of organics in the high-strength raw leachate imposed a high ozone demand.

Disinfection of a biologically treated and diluted leachate using NaOCl has also been investigated (Polprasert, 1977; Polprasert and Carlson, 1977). The effects of NaOCl dosage and hardness concentration were studied for bacterial and viral inactivation of a leachate seeded with E. coli and T-4 coliphage to increase bacterial and viral densities, respectively. A batch reactor was used to perform the bench-scale study, and dosages of 5 to 55 mg/l NaOCl for τ = 2 to 60 minutes were used. Greater than 99% bacterial inactivation was achieved for NaOCl dosages of 1 to 20 mg/l at a contact time of 30 minutes. The 5 mg/l NaOCl dosage was relatively ineffective, since only 90% bacterial inactivation occurred for a contact time of 60 minutes. Higher dosages were necessary for equivalent viral inactivation; 99% inactivation occurred with 48 mg/l NaOCl at a contact time of 60 minutes, and 99.99% inactivation occurred with 55 mg/l NaOCl at 60 minutes contact. Overall, a higher NaOCl dosage and a longer contact time were necessary for viral inactivation than for bacterial inactivation. Results at hardness concentrations of 250 to 1000 mg/l as $CaCO_3$ indicated that both bacterial and viral inactivation decreased as the hardness concentration increased.

Chemical Process Sludge Characteristics--

Chemical treatment with coagulants, precipitants, and oxidants generally did not achieve effective COD removal and chemical dosages were exceedingly high and not very practical. Moreover, large sludge volumes resulted as indicated in Figure 29. Sludge volumes greater than 5% of the original leachate volume were typical and were occasionally as high as 30 to 40%. Lime treatment produced the greatest sludge volume of all chemicals investigated, while the oxidants produced the smallest sludge volumes, typically 1%.

TABLE 26. BENCH-SCALE RESEARCH PERFORMED WITH CHEMICAL DISINFECTION OF LEACHATE

REFERENCE ITEM	(5,6,64)	(80,81)
Description of Study	Investigate use of O_3 for raw leachate to inactivate bacteria in a batch reactor.	Study bacterial and viral inactivation using NaOCl for biologically treated leachate using a batch reactor; effect of hardness; develop inactivation kinetic models.
Leachate Quality		
Bacterial Density, CFU/ml	300	$0.05\text{-}33 \times 10^7$ (seeded)
Viral Density, PFU/ml	-	$0.7\text{-}1.0 \times 10^7$ (seeded)
COD, mg/l	14,000	150
TOC, mg/l	5,200	-
NH_3-N, mg/l	-	1.8
Cr, mg/l	1.14	-
Cu, mg/l	0.39	-
Fe, mg/l	47	-
Pb, mg/l	0.025	-
Ni, mg/l	-	-
Zn, mg/l	12.5	-
Hardness, mg/l as $CaCO_3$	-	280-1000
pH	5.3	7.6
Disinfectant	O_3	NaOCl
Dosage, mg/l	10-163	5-55
Contact Time, minutes	-	2-60
Enumeration Technique		
Bacteria	Standard Plate Count.	Membrane filter
Virus	Not determined.	(Refer to Sobsey, *et al.*, 1974)
Conclusions	O_3 dosage of 110 mg/l at unknown contact time yielded leachate containing <30 CFU/ml.	Viral resistance to disinfection > bacterial resistance; hardness inhibited both bacterial and viral inactivation of NaOCl.

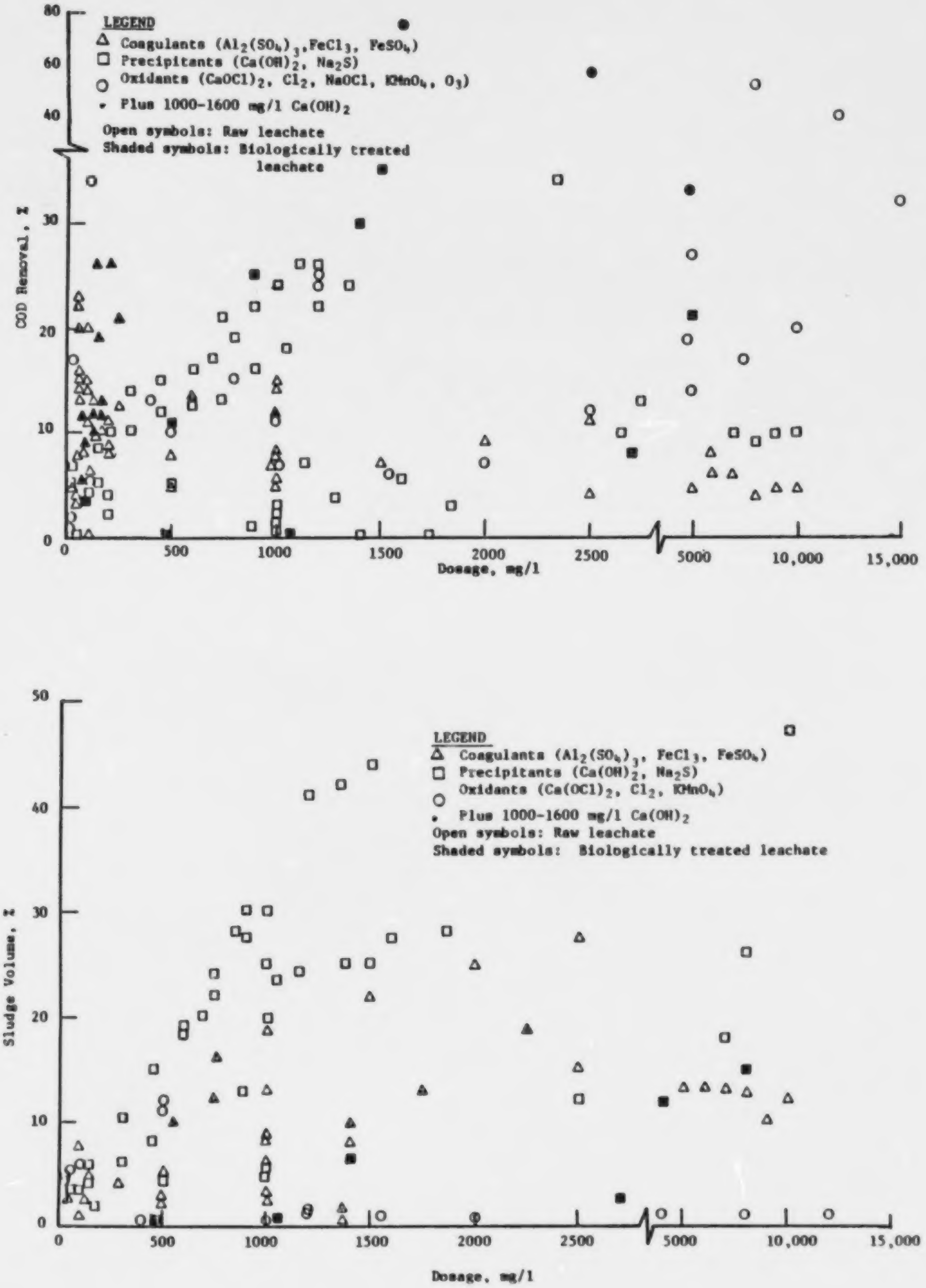

Figure 29. Comparison of the Organic Removal Efficiencies and Sludge Volumes Produced by the Application of Various Chemical Dosages to the Treatment of Leachates on Bench-Scale

Ionizing Radiation--

The application of ionizing radiation for the treatment of landfill leachates has been tested by one investigator (Yamazaki and Sawai, 1981). A medium-strength leachate (2000 mg/l TOC) was radiated with a 5-Kilocurie (KCi) ^{60}Co source which emitted an average dose of 0.6 mrad/hr. The effects of pH, aeration rate, and radiation dose on TOC removal were evaluated at room temperature. Maximum TOC removal (75%) was achieved at pH 4 and a radiation dose of 20 mrad/hr. At low radiation doses, aeration increases yielded increased TOC removals; these effects were much less noticeable at higher doses. As a result of the applied radiation, the organic compounds present in the leachate were converted from high molecular weight compounds to low molecular weight compounds. Humic and fulvic acid fractions were converted to low molecular weight carboxylic and phenolic compounds, alcohols, and other substances. Leachate biodegradability was believed to have improved as a result of the radiation, since it produced low molecular weight compounds. As such, this process may hold promise as a pre-treatment prior to more complete biological removal of organic constituents.

Ion Exchange--

Anionic and mixed ion exchange resins have been evaluated for polishing of biologically treated leachates as indicated in Appendix Table B-6 and summarized in Table 27. COD and TOC removals by ion exchange from these low-strength wastewaters ranged from 10 to 70% in both batch and continuous processes.

TABLE 27. SUMMARY OF ION EXCHANGE PERFORMANCE USING EFFLUENTS FROM AERATED LAGOON AND ACTIVATED SLUDGE LEACHATE TREATMENT SYSTEMS

				Influent Concentration mg/l		Removal, %		
Reference	Process	Leachate Type	pH	COD	TOC	COD	TOC	Comments
44,45,70	Anion Exchange	AL	8.8	500	200	6-59	26-43	Continuous
44,45,70	Anion Exchange	AL	6.2	500	200	48	43	Continuous
205,207	Anion Exchange	AS	5.0-7.7	180	-	68-36	-	Batch;2-10 g/l
205,207	Anion and Cation Exchange	AS	7.3	185	-	10	-	Batch;2-10 g/l

Cationic ion exchange has also been studied for the removal of metals from leachates using glauconitic greensand (GG) a common geological stratum indigenous to the Delaware and New Jersey regions of the United States and reported as having significant cation exchange capacity (Spoljaric and Crawford, 1979 a,b). The research focused on the effects of flow rate on metals removal in a continuous flow processes with flow rates of 0.1 and 1.0 l/min. As shown in Table 28, the lower flow rate provided better removal copper, lead and nickel (96% or greater) over iron (86%) and zinc (67%). This is in contrast to the other treatment processes (biological and chemical addition studies) where iron and zinc were typically most affected and lead and nickel were least affected. Chromium, manganese, calcium and magnesium were fairly well removed at the lower flow rate, whereas, potassium and sodium were poorly removed. Although the cation exchange capacity of GG is low (2.1-3.6 meq/100 g), this process could be economical depending on handling costs for the exchange media.

TABLE 28. SUMMARY OF GLAUCONITIC GREENSAND (GG) PERFORMANCE FOR THE REMOVAL OF METALS FROM LEACHATE

Reference	Parameter	Influent Concentration, mg/l		Removal, %		Comments
		0.1 l/min	1.0 l/min	0.1 l/min	1.0 l/min	
29,170	Cd	0.006	0.08	83	96	Continuous flow, upflow sand filter bed. Lower flow rate provided better removal; exchange capacity on the order of 2.1-3.6 meq/100g.
	Ca	129	181	63	22	
	Cr	0.03	0.13	66	-	
	Cu	0.38	0.28	99	14	
	Fe	8.1	14.0	86	3	
	Pb	0.13	0.18	99	13	
	Mg	62	164	67	26	
	Mn	4.1	6.1	88	48	
	Ni	0.07	0.21	96	14	
	K	122	364	39	62	
	Na	275	585	36	0	
	Zn	0.49	0.78	67	20	
	pH	7.7-6.3	7.5-6.6			

Adsorption--

The adsorption of organics and metals from leachates has been studied using activated carbon and peat. Activated carbon has received the vast majority of study, having been evaluated in batch and continuous processes with granular and powdered carbons. The evaluations have generally involved the use of biologically or chemically treated leachates. The biological treatment effluents were typified by COD concentrations ranging from 200 to 800 mg/l, whereas, the chemical treatment effluents generally contained 2000 to 3000 mg/l COD.

As indicated in Appendix Table B-7, activated carbon was generally capable of removing 30 to 70% of the residual COD and TOC at retention times of 1 to 15 minutes in continuous flow processes. Removal efficiencies were lower for chemically treated leachates than for biologically treated leachates. Further comparison of these wastewaters is provided in Figure 30 by Freundlich isotherms for the batch adsorption studies listed in Appendix Table B-7. The COD and TOC based isotherms shown in the figure have steep slopes, suggesting that continuous operation would be more efficient than batch adsorption. Although limited data were available, the biologically and chemically treated leachate isotherms were fairly similar, having similar relationships between the equilibrium organic concentration and the adsorptive capacity of the carbon. The raw leachate isotherms deviated from the other two, due mainly to the higher concentrations of organics imposed, but also to some degree to differences in organic composition. A summary of the Freundlich isotherm parameters derived from each case is provided in Table 29.

TABLE 29. SUMMARY OF FREUNDLICH ISOTHERM PARAMETERS FOR BENCH-SCALE ACTIVATED CARBON ADSORPTION OF RAW LEACHATE AND TREATED LEACHATE

References	Leachate Type	$\frac{\text{COD}}{\text{TOC}}$	C_o, mg/l		X/M, $\frac{\text{mg COD or TOC}}{\text{mg AC}}$		1/n, $\frac{\text{mg COD or TOC}}{\text{mg AC·mg/l}}$	
			COD	TOC	COD	TOC	COD	TOC
45,134,151	Raw	-	5000	395-13,000	2.5	0.046-0.30	9.5	0.6-1.2
45,205	Biologically Treated	2.4-3.8	184-830	210-320	0.261-0.54	0.102-0.74	0.7-2.3	-
28,143	Chemically Treated	3.3-3.7	508-2990	153-150	0.20-0.80	0.14-0.165	1.4-3.2	0.97-1.1
143	Biologically Plus Chemically Treated	3.0-3.7	192-344	130-230	0.15-0.66	0.13-0.23	0.98-5.9	2.4-2.9

$X/M = KC_o^{1/n}$

-Data not given.

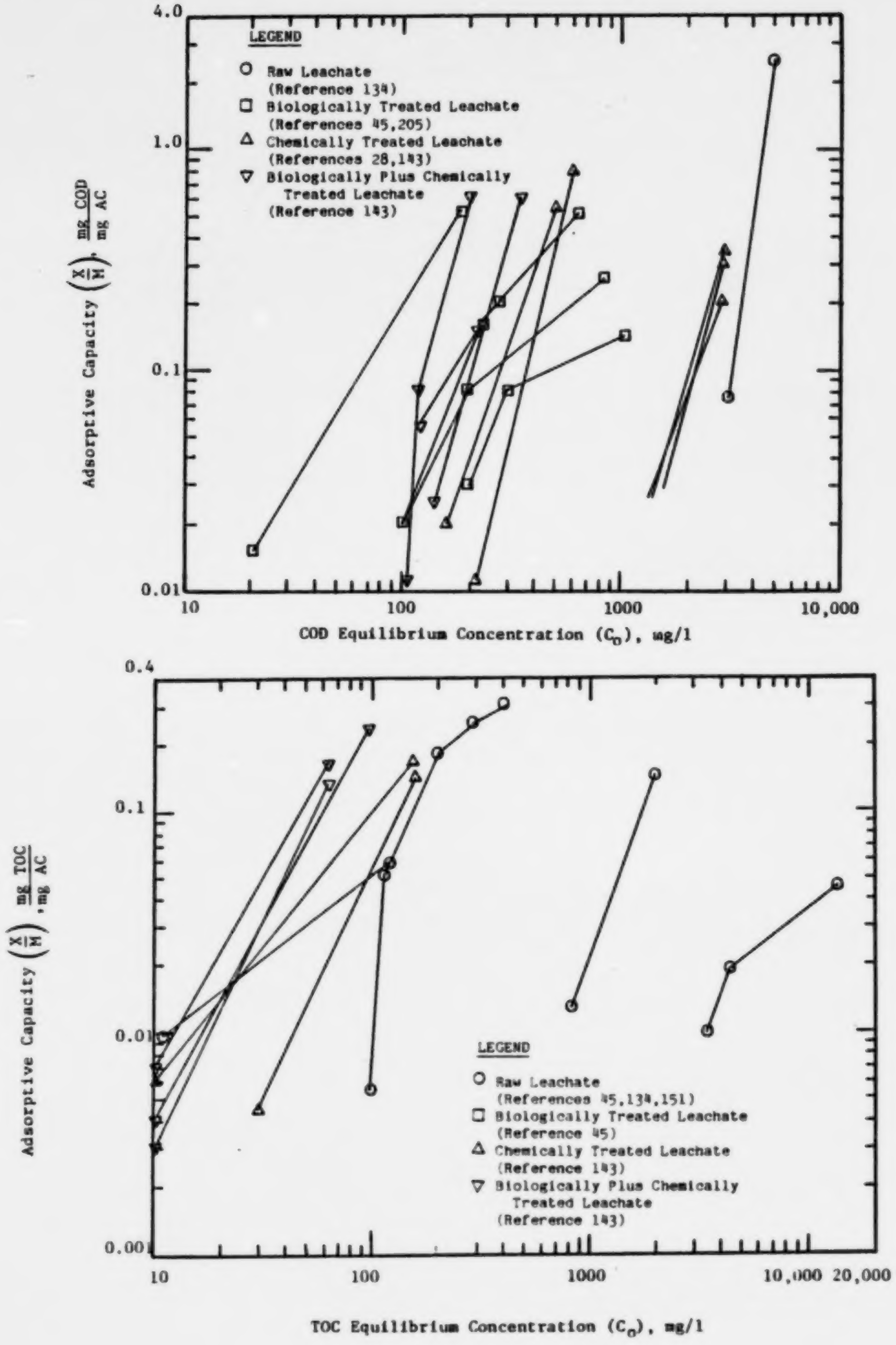

Figure 30. Freundlich Isotherm Curves for Bench-Scale Batch Activated Carbon Treatment of Raw, Biologically or Chemically Treated Leachates

Peat adsorption studies (Lidkea, 1974; Corbett, 1975; Cameron, 1978) on organic and metals removals from leachate are summarized in Table 30. Continuous flow columns filled with dried peat were used to evaluate the effects of pH on process performance. Alkaline conditions were more effective than were acidic conditions. At pH 7.1 to 7.8, the peat columns removed 86% COD, 95% NH_3-N and greater than 90% of all metals studied except lead. Metals removal was attributed to a combined precipitation/filtration mechanism at the alkaline pH values.

TABLE 30. SUMMARY OF THE PERFORMANCE OF PEAT FOR ADSORPTION OF ORGANICS AND METALS FROM LEACHATE

References	Parameter	Leachate Concentration, mg/l		Removal, %		Comments
32, 56, 170	pH	4.8	7.1	4.8	7.1	Continous upflow column using dried peat as an adsorption media
	COD	830	830	66	66	
	TKN	-	465	-	95	
	Ca	254	174	66	92	
	Fe	27	22	82	99	
	Pb	0.03	0.06	98	73	
	Mg	106	126	55	96	
	Mn	0.52	0.61	67	92	
	K	580	126	71	96	
	Na	1400	780	70	95	
	Zn	0.43	0.60	47	90	

Metals Removal--

A summary of heavy metal and alkali and alkaline earth metals removals achieved by the ion exchange and adsorption processes is provided in Table 31. Due to the limited data available, definitive statements are not possible, although the ion exchange appeared to be superior to adsorption for the removal of both heavy and alkaline earth metals.

As indicated in Appendix Table B-8, activated carbon was successful in removing 96% of the iron from raw and ozonated leachates; the performance achieved seemed dependent on carbon dosage. Using batch adsorption tests, an 8 g/l dose of powdered activated carbon (PAC) improved iron removal from 73%

at a 2 g/l dose to 96% (Ho, et al., 1974). Further increases in carbon dosage yielded little improvement (97% iron removal at 16 g/l PAC dose). Data provided in the literature for other metals were insufficient for comparison.

TABLE 31. SUMMARY OF HEAVY METAL AND ALKALI AND ALKALINE EARTH METAL REMOVAL DATA WITH ACTIVATED CARBON ADSORPTION AND RESIN ION-EXCHANGE TREATMENT OF LEACHATE

Heavy Metals

	Cd	Cr	Cu	Fe	Pb	Mn	Ni	Zn
Concentration Range, mg/l	0.03-0.08	0.07-0.13	0.24-0.28	14-66	0.18-0.23	6.1-25	0.13-60	0.7-60
Removal Range, %	27-96	0	0-14	10-97	22-33	21-87	0-37	0-99

Alkali and Alkaline Earth Metals

	Ca	Mg	K	Na
Concentration Range, mg/l	15-181	15-164	63-380	200-585
Removal Range, %	0-95	0-99	0-95	0-99

[*] Not applicable. Not sufficient data for true statistical average.
[**] Activated Carbon (AC); Ion Exchange (IX)
-Data not given.

The removal of alkaline earth metals from raw leachates was also somewhat varied and, although limited data were available, it appeared that ion exchange offered better removal than adsorption. A batch activated carbon study of metals removal (Karr, 1972) indicated that manganese was best removed, but it was also present in lowest concentration as indicated in Appendix Table B-9. Calcium, magnesium, and potassium, present in higher concentrations, were removed by 40% or less. The ion exchange processes were more successful in removing these constituents, typically exhibiting 75 to 95% calcium, >95% magnesium, 50 to 95% potassium, and up to 99% sodium removals for biologically treated leachates, depending on the resin type and dosage applied.

A comparison of ion exchange (IX) and activated carbon (AC) treatment of raw leachate is also presented in Appendix Table B-9. The limited data presented seem to suggest that glauconitic greensand is superior to AC at similar flow rates and bed volumes. At lower flow rates, the superiority of IX becomes increasingly evident.

Reverse Osmosis--

Reverse osmosis has received consideration as both an initial (raw leachate) treatment step and a final polishing step (using biological, AC or IX treatment process effluents). Raw leachates were initially studied (Chian and DeWalle, 1977b) and, as summarized in Table 32, reverse osmosis (RO) was fairly efficient in removing the majority of the residual TOC. Two types of RO membranes were used, each having different polar characteristics. The more polar membrane (NS-100) achieved slightly superior TOC removal than the cellulose acetate (KP-98) membrane at both pH 5.5 and 8.0, although the difference in performance was much more marked at pH 5.5.

The major problem associated with RO treatment of raw leachates was membrane fouling due to solids, colloidal material, and iron hydroxides. Therefore, emphasis was also placed on the removal of TOC from aerated lagoon, activated carbon, and ion exchange process effluents using RO as summarized in Table 32.

TABLE 32. SUMMARY OF REVERSE OSMOSIS PERFORMANCE FOR THE REMOVAL OF COD FROM RAW AND BIOLOGICALLY TREATED LANDFILL LEACHATES

References	Process	Leachate Type	pH	Influent COD, mg/l	COD Removal, %	Operating Conditions*
44,45,70	Reverse Osmosis	Raw	5.5	13,000-18,500	85-98	P=600,1500 psi, (KP-98)
		Raw	5.5	13,000-18,500	98-99	P=600,1500 psi, (NS-100)
		AL	8.8	214	95	P=600 psi
		AC	8.8	48	86	P=600 psi
		IX	5.5	119-143	94-97	P=600 psi

*1 psi = 6.895 kN/m^2

AS = Activated sludge effluent
AL = Aerated lagoon effluent
AC = Activated carbon effluent
IX = Anion exchange effluent

Only the NS-100 membrane was utilized for the treated leachate tests, since it was found to be superior with raw leachate. Application of RO to the activated carbon treatment effluent was the least successful, achieving only 86% TOC removal as compared to 94 to 96% removals for aerated lagoon and ion exchange treatment effluents. Although successful as an effluent polishing measure by itself, the problem of membrane fouling was considered serious enough to warrant filtration or coagulation of the treatment effluents prior to RO polishing (Chian and DeWalle, 1977b).

Full-Scale Physical/Chemical Leachate Treatment

Chemical treatment using coagulants and precipitants, NH_3 stripping, and activated carbon adsorption have been tested at several full-scale leachate treatment facilities. These studies are summarized in Table 33 along with a process description and location of the landfill and treatment facility. All of the landfills were classified as municipal solid waste landfills except for the Love Canal landfill (McDougall and Fusco, 1980; McDougall, et al., 1980).

TABLE 33. FULL-SCALE LEACHATE TREATMENT FACILITIES USING A PHYSICAL/CHEMICAL PROCESS

Reference	Process	Process Description	Location
26	Precipitation	Lime addition for heavy metal removal.	North Hempstead, New York
133	Coagulation and Adsorption	Alum and polymer addition for pretreatment prior to AC adsorption for organic and heavy metal removal.	Franklin County, Pennsylvania
178, 179	Adsorption	NaOH addition for pretreatment prior to AC adsorption for removal of toxic organics, most classified as priority pollutants.	Love Canal, New York
231	Chemical Addition	Chemical addition prior to treatment by aerated lagoon and activated sludge.	Pennsylvania (2 landfills)
245-248	Precipitation; NH_3 Stripping; Nuetralization	Lime addition for heavy metal removal and to raise pH; Air stripping of NH_3 at alkaline pH using a lagoon; Sulfuric and phosphoric acid addition for neutralization.	Bucks County, Pennsylvania

Precipitation/Coagulation--

Chemical addition has been the most common full-scale physical/chemical process used for landfill leachate treatment. A summary of the treatment performance and design parameters for the full-scale treatment facilities using this approach is included in Table 34. The available information has been separated into influent and effluent quality, pretreatment, treatment, and sludge characteristics.

TABLE 34. SUMMARY OF PERFORMANCE AND DESIGN PARAMETERS FOR FULL-SCALE PHYSICAL/CHEMICAL LEACHATE TREATMENT FACILITIES

ITEM \ REFERENCE	(26)	(133)	(173,179)	(231)	(245-248)		
Process	Precipitation	Coagulation Adsorption	Adsorption	Chemical Addition	Precipitation NH_3 Stripping S1*	Precipitation S2*	Precipitation NH_3 Stripping S2*
Influent Quality							
BOD_5, mg/l	10,000	100	-	-	11,900	10,400	11,700
COD, mg/l	14,000	-	11,500	2500	18,500	16,600	18,600
TOC, mg/l	-	-	4300	-	-	-	-
TKN, mg/l	700	-	-	90	760	1170	785
NH_3-N, mg/l	600	10	-	50	760	1170	785
TDS, mg/l	-	-	-	-	13,500	12,700	10,500
Cd, mg/l	0.05	-	-	-	0.08	0.07	0.09
Cr, mg/l	-	-	-	-	0.26	0.25	0.25
Cu, mg/l	-	0.56	-	-	0.40	0.46	0.43
Fe, mg/l	1000	20	330	750	333	350	300
Pb, mg/l	-	0.10	0.4	-	0.74	0.75	0.68
Ni, mg/l	-	-	-	-	1.76	1.58	1.60
Zn, mg/l	8	-	-	20	20	19	16
pH	6.0	7.6	5.6	6.0	6.7	6.9	6.9
Q, m^3/day	136;245	71	65	549	79.5	86.3	86.3
Pretreatment	Preaeration	No	Caustic Addition, multi-media filtration	No	No	No	No
Treatment							
Coagulation-Flocculation			-	-			
Dosage, mg/l	1650,Lime	-,Alum			3000,Lime	3600,Lime	2300,Lime
τ,minutes	15-30	-			-	-	-
pH	-	-			-	9.0-11.7	9.0-11.7
Settling			-	-			
τ,hours	3;1.7	4			11	10	10
Overflow rate, m^3/m^2·day	20;37	15			7.6	8.2	8.2
NH_3 Stripping	No	No	No	No		No	
τ,days					12		11
pH					10		10

TABLE 34 (Continued)

AC Adsorption	No		2-9070 Kg GAC units in series	No	No	No	No
τ,min		3	-				
X/M, mg TOC/g AC		-	200				
Effluent Quality							
BOD_5, mg/l	-	28(72)	-	-	3930(67)	5270(49)	3600(69)
COD, mg/l (%R)	-	-	-	200(92)	6890(63)	7200(57)	8800(53)
TOC, mg/l (%R)	-	-	100(98)	-	-	-	-
TKN, mg/l (%R)	-	-	-	20(78)	350(54)	890(24)	410(48)
NH_3-N, mg/l (%R)	-	1.7(83)	-	15(70)	350(54)	890(24)	410(48)
TDS, mg/l (%R)	-	-	-	-	6000(56)	7970(37)	4650(56)
Cd, mg/l (%R)	-	-	-	-	0.03(60)	0.03(60)	0.04(60)
Cr, mg/l (%R)	-	-	-	-	0.07(70)	0.09(60)	0.08(70)
Cu, mg/l (%R)	-	-	-	-	0.31(23)	0.10(80)	0.27(37)
Fe, mg/l (%R)	2.3(>99); 12(99)	-	-	5(99)	3.2(99)	4(99)	6(98)
Pb, mg/l (%R)	-	0(>90)	-	-	0.17(77)	0.24(68)	0.23(66)
Ni, mg/l (%R)	-	-	-	-	0.61(65)	0.57(64)	0.73(54)
Zn, mg/l (%R)	0.03(>99); 0.03(>99)	-	-	2(90)	0.6(97)	0.6(97)	0.9(94)
pH	-	7.4	-	7.5	8.6	8.5	8.7
Sludge Characteristics	1230 kg/day generated	-	-	-	-	-	-
Effluent Disposal	-	Surface water	POTW	Surface water	Surface water or spray irrigation		

*S1 = System 1; S2 = System 2

Lime was the only precipitant used for organic and metals removal. Very high lime doses of 2300 to 3600 mg/l were necessary to achieve about 50 to 70% BOD_5 and COD removal. As with the bench-scale processes, the removal of heavy metals was significant, especially in the case of Fe and Zn where 98 to >99% removal was achieved at influent concentrations of 300 to 1000 mg/l Fe and 8 to 20 mg/l Zn. The other heavy metals, Cd, Cr, Cu, Pb and Ni, were also removed, but influent concentrations were typically less than 1 mg/l and a correspondingly high removal efficiency would not be anticipated. Although the effectiveness of lime in decreasing heavy metal concentrations at the full-scale treatment operations was similar to that for the bench-scale studies, less than 40% COD removal was generally achieved in the bench-scale studies. Greater COD removal was achieved on full-scale, most likely as a result of prior NH_3 stripping which also promoted the removal of volatile organics. As indicated in Table 34, the NH_3 stripping step was performed at pH 10 in a lagoon having a detention time of 11 to 12 days.

Alum was also used in one full-scale treatment facility to treat a low-strength leachate characterized by a BOD_5 concentration of 100 mg/l (Hemsley and Koster, 1980). This facility was able to achieve about 70% BOD_5 removal, but the alum dosage was not reported. Additional BOD_5 removal was achieved with AC adsorption following alum coagulation of the leachate. However, these two processes were not separately monitored and their individual removal contributions were not noted.

Ammonia Stripping--

Ammonia stripping has been attempted at one landfill under two different treatment conditions as indicated in Table 34. The leachate was pretreated with lime to raise the pH to about 10 and air stripping was then applied for NH_3 removal. The stripping process occurred in a large lagoon having a detention time of 11 to 12 days. Ammonia nitrogen removal for the two conditions ranged from 48 to 54% with influent NH_3-N concentrations of 760 to 785 mg/l. Given the long detention time used, NH_3-N removal by stripping was not as promising as would be expected. Operational problems with pH control might have been the cause of the relatively poor stripping efficiency.

Activated Carbon Adsorption--

Activated carbon adsorption was applied at two landfills for polishing following alum coagulation (Hemsley and Koster, 1980). Approximately 70% BOD_5 removal was achieved by this treatment process when the influent concentration was 100 mg/l BOD_5.

The use of AC adsorption of leachate produced from a landfill used primarily for the disposal of organic chemicals has also been reported (McDougall and Fusco, 1980; McDougall, et al., 1980). Following caustic addition and multi-media filtration, two granular activated carbon adsorption units were used to polish the effluent prior to discharge to a publically owned treatment work (POTW). The adsorption process was found to be 98% efficient in TOC removal for an influent TOC concentration of 4300 mg/l. The maximum adsorptive capacity was 200 mg TOC/g AC and the treated effluent contained about 100 mg/l TOC and priority pollutant concentrations that were typically below detectable limits.

FINAL LEACHATE DISPOSAL

Following treatment of leachates by any of the previously discussed processes, ultimate disposal in an environmentally sound manner will be required. Options available for ultimate disposal include land application, discharge to surface waters, and discharge to a publicly owned treatment works.

Land Disposal

Land application of treated leachates has been tested on full- and bench-scale. Full-scale land applications by spray irrigation and ridge-and-furrow methods have been reported as indicated in Table 35. Unfortunately, the capabilities of these applications for final pollutant attenuation could not be ascertained, since groundwater quality was not monitored and soil characteristics at each land application site were not revealed. From a hydrologic perspective, the application rates used were apparently acceptable, since problems associated with over-application (such as flooding) were not reported.

TABLE 35. EFFLUENT DISPOSAL PRACTICES EMPLOYED BY FULL-SCALE LEACHATE TREATMENT FACILITIES

		Land Application		Leachate Quality Prior To Disposal		
Reference	Disposal Method*	Flow, m^3/day	Rate, $l/m^2 \cdot day$	BOD_5, mg/l	COD, mg/l	pH
15	SW	355	NA	10	-	7.5
15	SW	45.4	NA	920	-	7.4
26	POTW	303	NA	-	-	-
113-115	Ridge and furrow	39	4.7	<800	<1500	-
166	SW	77.8	NA	10	120	7.3
187	SI	150	9.4	1200	2280	-
231	SW	549	NA	100	-	-
244	SI	13	10	25	-	-
245-248	SI	39-78	0.37	120-2150	940-4650	7.6-8.6
	SW	39-78	NA	120-2150	940-4650	7.6-8.6

*POTW = Discharge to publicly owned treatment works
SI = Spray irrigation
SW = Surface water discharge
- = Data not given
NA = Not Applicable

Discharge to POTW

One alternative for the ultimate disposal of treated leachates is the discharge to publicly owned treatment works (POTW). This practice must also be evaluated on a site-specific basis, since leachate quantity and quality may affect the performance of the POTW. Data from one landfill utilizing this discharge method are indicated in Table 35, although no data on leachate quality were given. The leachate apparently posed no detrimental effects on the quality of the effluent from the POTW.

More data were available from a bench-scale study designed to simulate the spray irrigation process (Chan, et al., 1978). A test column was constructed and filled with native soil from the landfill site. Lime treated leachate was then applied at 37 $l/m^2 \cdot day$ or at a loading rate chosen to stimulate conditions planned for full-scale operation. The lime treated leachate was characterized by 5400 mg/l COD, 690 mg/l Na, 540 mg/l K, 600 mg/l Ca, 104 mg/l Mg, and a pH value of 10. Divalent cations were better attenuated in the soil (comprised of 12% clay) than the monovalent cations, and complete COD breakthrough occurred in less than three bed volumes. Consequently, land application would be better practiced for lower-strength leachates. More research is needed on the fate of pollutants in actual leachate land spreading settings.

Surface Water Discharge

Discharge of landfill leachates to surface waters is subject to the same restrictions as applied to any point source wastewater. Accordingly, the quality of leachate required prior to surface water discharge is dictated by a number of site-specific technical and regulatory factors, including the assimilative capacity of the receiving water. If leachate quality exceeds recommended limitations, alternative disposal options must be sought. No data were available in the literature on the use of direct discharge of untreated leachates for ultimate disposal.

SECTION 7

GAS MANAGEMENT

GENERAL PERSPECTIVE

The release of gases by biological activity or by evaporation (volatilization) of waste constituents may pose certain hazards to landfill operators and/or nearby residents. As previously outlined, the most obvious of these hazards include the potential for fires and explosions. The control of hazards has led to the development of various strategies for landfill gas control and an emphasis on gas collection and energy recovery. Accordingly, the state-of-the-art in landfill gas management includes an integration of the elements of landfill lining (containment) with gas collection, treatment and possible power generation. Although the latter subject was considered beyond the scope of this report, the technology associated with landfill gas-fired electrical generation is essentially identical to that associated with other fuel sources and is generally on-the-shelf and available from a number of manufacturers. Similarly, liner technology has been addressed elsewhere, and has been the subject of several recent review publications (Landreth, 1980; EPA, 1983; National Sanitation Foundation, 1983).

The purpose of this section is to present an overview of literature pertinent to factors affecting gas production and a summary of reported gas yields, composition and production rates associated with various landfill operations. In addition, gas collection and treatment technologies (for both on-site generation and pipeline uses) will be briefly introduced in somewhat less detail than presented in Section 4, since these subjects have been comprehensively addressed by others (EPA, 1979; EMCON, 1980; DOE, 1981; Halvadakis, et al., 1983). Moreover, attempts at providing updates on full-scale operations were hampered by the brevity, lack of data and the presumptive nature of many of the reports constituting the available literature.

GAS PRODUCTION

The sizing and implementation of gas handling equipment requires a prediction of gas production rates, yields, and gas composition from a particular landfill setting. Such a prediction may be based on theory or formulated from comparisons with empirical results from published laboratory and field experiences. In either case, an understanding of the biochemical and physical factors affecting gas production and of site conditions is necessary. In particular, the phasic nature of landfill stabilization (SECTION 3) and the corresponding biophysical variations must be coupled to the refuse placement and leachate control technologies being utilized. Integration of time-dependent gas quantity and quality expectations (Figure 1, Table 3) with refuse placement schedules may provide for a redundant use of both gas and leachate handling equipment, particularly where leachate recycle is being implemented. Therefore, the following briefly summarizes the factors affecting gas production in landfills, with an emphasis on methanogenesis (Phase IV, Figure 1). Theoretical gas yield models are then reviewed, followed by a summary of gas production rates, compositions and total methane yields reported in the literature.

Factors Affecting Landfill Gas Production

Gas production in landfills is affected by many variables, including the nature of wastes placed, moisture content, particle size and degree of refuse compaction, buffer capacity, nutrient sufficiency, temperature, and the gas extraction method. These factors have been reviewed in detail by Rees (1980) and Halvadakis, et al., (1983). From these and other sources, the following general conclusions may be offered regarding the influence of these variables on gas production.

Nature of Refuse Placed--

As reviewed previously, the sources of solid waste placed in a sanitary landfill are largely a function of location and may vary considerably according to residential, commercial or industrial origin. The nature of these wastes influences the potential for gas production in terms of: 1) the relative availability of a usable substrate, including its organic, moisture and nutrient contents; 2) the presence of potential inhibitors; and, 3) the formation of localized "micro environments" which may be isolated from the overall liquid or gaseous transport phases. As indicated in Table 2, paper products are a major contributor to the overall composition of refuse, although these are generally more resistant to biodegradation than food and most garden wastes. Industrial wastes are important with regard to the buffers and metallic and other constituents they provide and may impart either benefical or detrimental influences depending on their relative magnitudes and propensity for reaction.

Moisture Content--

Water or moisture (leachate) provides the transport phase for organic substrates and nutrients and is also instrumental in establishing the anaerobic environment needed for methane production. Up to a point, increasing the moisture content increases the rate of methane production and the ultimate methane yield. In general, it may be expected that methane production rates will increase with increasing moisture up to approximately 60% (40% solids), with higher moisture imparting neither an increase nor a decrease in the maximum gas production rate.

Eliassen (1975) considered the moisture content requisite for biological decomposition and reported optimum moisture ranges of 50 to 70% and 30 to 80% for new and older landfills, respectively. Chian and DeWalle (1979) reported that 75% moisture content or above was best for biodegradation of municipal solid waste, although the presence of more water was also recognized as resulting in production of larger quantities of leachate requiring treatment. In spite of these observations, the large number of interrelated variables involved in these studies has precluded a clear determination of moisture effects; uniformity of moisture is probably equally important as quantity of moisture, as demonstrated to some degree by leachate recirculation studies.

Particle Size and Degree of Refuse Compaction--

Particle size reduction by refuse shredding may be expected to increase gas production rates by increasing the surface area available for leaching and/or biological activity, and by improving the ability to retain moisture (DeWalle, et al., 1978; Fungaroli 1979), although Buivid (1980) reported contrary results. Therefore, none of the results of these studies are clearly conclusive, primarily due to the wide number of variables involved.

Literature data on refuse density and/or effects of compaction are likewise inconclusive. Compaction will tend to optimize the volume of waste which can be placed in a given landfill volume. However, compaction may be expected to impede moisture and gas flow through the wastes, thereby increasing the potential for microenvironment formation and leading to decreased refuse stabilization or methane release rates. Therefore, more focused and systematic studies are needed on both of these operational variables.

Buffer Capacity--

Buffer addition has been repeatedly demonstrated as beneficial to accelerating biological stabilization and increasing gas production rates (Pohland, 1980; Pacey, 1983). Sufficient buffer is needed to moderate the effects of volatile acids and other acid products which tend to depress the pH below the desired level for methanogenesis (pH 6.6-7.4). As yet, no systematic studies of specific buffer additions to landfills have been performed. The practice of buffer addition is expected to be quantitatively linked to site specific variables. Therefore, the approach to buffer addition could be based on leachate analysis and application during leachate recycle or by injection, or on anticipated need and augmentation of the refuse as it is being placed. Addition of digested sewage sludge to the refuse during landfilling is an example of the latter approach.

Nutrients--

The same considerations mentioned for buffer applications apply for nutrient additions. Nutrient sufficiency may be best assured through initial addition or after leachate analysis by augmentation as needed again through leachate recycle or injection. Municipal solid wastes generally contain the nutrients necessary for effective biological conversion, although Pohland (1975) has shown that phosphorus may become limiting during the latter stages of biostabilization. Nutrient additions to simulated landfill cells have not produced distinguishable effects, again due to other operational differences and, in particular, the fairly common practice of adding microbial seed along with nutrients. If control over stabilization rates and gas production are considered crucial, the issue of nutrient sufficiency should again receive more systematic study.

Temperature--

Temperature affects microbial activity within landfills and vice versa. In the upper aerobic layers (1 to 2 m), temperatures may range from 50 to 70°C, whereas, at lower aerobic levels (2-3 m), temperatures generally range from 25 to 40°C. Following the depletion of oxygen and the change from aerobic to anaerobic metabolism, temperatures within the landfill will decrease and remain moderated by ambient conditions. Rees (1980) reported on a method of landfill temperature moderation by utilizing a refuse placement strategy which takes advantage of aerobic biological heat generation. Fresh wastes were placed in areas adjacent to regions of active methanogenic stabilization to promote accelerated conversion made possible at the higher temperatures.

Gas Extraction--

The withdrawal of landfill gases at rates higher than their biological production will lead to the introduction of air into the landfill. This may not only inhibit the methanogens, but lead to excessive quantities of nitrogen and oxygen in the product gas. The latter consequence would correspondingly decrease the overall energy value of the gas and require otherwise unnecessary and expensive gas treatment. (There have been undocumented reports of reduced methane generation rates of landfill sites operated with gas extraction facilities.)

Gas Yield Projections

Ultimate gas (methane) yields are important in determining the economic feasibility of gas recovery projects. However, they are not very useful in sizing recovery equipment unless coupled to a prediction of measurement of gas yields. Several methods are available for formulating gas yields, including both theoretical and empirical approaches. These are reviewed in more detail by EMCON (1980) and Halvadakis, *et al*. (1983) and are briefly summarized here.

Theoretical Models--

Stoichiometric Methods--A number of investigators have derived gas production estimates by making assumptions on the chemical composition of municipal solid wastes (MSW) and applying these assumptions to the Buswell equation for methanogenesis. This analysis may be performed using the entire MSW content or by making assumptions about biodegradabilities of the major waste fractions, e.g., food and garden wastes, papers, textiles, wood, leather, etc. In performing such an analysis, chemical formulas for MSW listed in Table 36 are combined with Equations 2 or 3 below, either using a formula for the overall MSW or a summation of yields from its individual components. The number of moles of each compound can then be calculated based upon the quantity of wastes handled, and the equations can also be used to determine the resultant moles or volumes of gas to be expected upon conversion of the waste.

Buswell equation:

$$C_nH_aO_b + \left(n - \frac{a}{4} - \frac{b}{2}\right)H_2O \rightarrow \left(\frac{n}{2} - \frac{a}{8} + \frac{b}{4}\right)CO_2 + \left(\frac{n}{2} + \frac{a}{8} - \frac{b}{4}\right)CH_4 \qquad \ldots(2)$$

Modified Buswell equation (Mao and Pohland, 1973):

$$C_aH_bO_cN_dS_e + \left(a - \frac{b}{4} - \frac{c}{2} + \frac{3d}{4} + \frac{e}{2}\right)H_2O \rightarrow \left(\frac{a}{2} + \frac{b}{8} - \frac{c}{4} - \frac{3d}{8} - \frac{e}{4}\right)CH_4 +$$

$$\left(\frac{a}{2} - \frac{b}{8} + \frac{c}{4} + \frac{3d}{8} + \frac{e}{4}\right)CO_2 + dNH_3 + eH_2S \qquad \ldots(3)$$

TABLE 36. EXAMPLES OF MUNICIPAL SOLID WASTE CHEMICAL FORMULAS APPLIED TO THEORETICAL METHANE YIELD MODELS*

Waste Component	Chemical Formula
Municipal Solid Waste	$C_{99}H_{149}O_{59}N$
Paper, Garden Wastes, Wood	$C_{203}H_{334}O_{138}N$
Food Wastes	$C_{16}H_{27}O_8N$
Cellulose	$C_6H_{10}O_5$

*Adopted from EMCON, 1980.

Examples of these calculations as well as assumptions of biodegradability and weight fractions are reviewed by EPA (1979) and EMCON (1980). A summary of theoretical gas yields predicted by several authors is given in Table 37.

TABLE 37. SUMMARY OF THEORETICAL GAS YIELDS FROM MUNICIPAL SOLID WASTE REPORTED IN THE LITERATURE

Reference	Method	Total Gas Yield Prediction, m^3/kg Dry Waste	Methane Yield Prediction, m^3/kg Dry Waste
5	MSW (Overall)	0.41	0.24
4	MSW (Overall)	0.42	0.21
86	MSW (Overall)	0.46	0.25
21	MSW (Overall)	0.45	0.23
116	Weighted Biodegradability	0.35	0.17
61	Weighted Biodegradability	0.19	0.09
199	Weighted Biodegradability	0.25	0.12
191	Weighted Biodegradability	0.12	0.06

These methods and the yields summarized in Table 37 are, at best, rough estimates of the potential gas production from landfill biodegradation of organic refuse constituents. As demonstrated below, they fail to include the influences of numerous factors such as the extent of aerobic and anaerobic decomposition, nutrient limitations, biological inhibition, and physical-chemical interactions which will generally serve to decrease the predicted methane yields. Moreover, these assumptions project a 100% recovery of gases produced, which on full-scale is impractical due to the high potential for uncontrolled gas migration, escape and entrapment.

Empirical Gas Yield Projections

Field and laboratory observations serve as the best indicator of actual gas yields from sanitary landfills. Gas yields reported in the literature for lysimeter and field studies are summarized in Table 38. As shown in the table, gas yields reported for small lysimeters were generally higher than those reported for larger landfill simulators. Although these results may be expected due to the greater potential on full-scale for localization of activity (microenvironment isolation), gas entrapment and leaks, moisture short-circuiting, etc., the data available to date are insufficient to quantify these factors. Therefore, gas yields reported for lysimeters should be used with caution when extrapolating for full-scale predictions. Data from full-scale operations would be the best indicator, but availability of such data is still limited. Moreover, older landfills which may have reached maturation have not been routinely examined with respect to refuse characteristics and/or gas yields. Newer landfills have yet to reach maturation so that even with routine analysis, total gas yields cannot be formulated and/or substantiated. Such data acquisition is also impeded by the variety and inherent uncertainty of gas collection methods employed at various sites. This problem is further magnified by a lack of understanding of the biochemical interactions occurring within the landfill and the absence of uniform and reliable data collection protocols.

The experimental data presented in Table 38 confirm the impracticality of utilizing theoretical predictions of gas yield. The yields determined experimentally were generally on the order of 10% (or less) of the theoretical predictions presented in Table 37.

Gas Production Rate Predictions

Several authors have developed mathematical models in attempts to describe gas production rates at landfills (see review by EMCON, 1980). However, these models are basically curve fitting techniques for which sufficient data are presently not available. Therefore, current gas production rate predictions are generally obtained by comparing overall gas yields from laboratory studies to the total "stabilization" time, by installing observation wells (EMCON, 1980; DOE, 1981), or by literature comparison. A summary of gas production rate data reported in the literature is presented in Table 39 for small-scale studies, and in Table 40 for full-scale operations.

The variations in lab-scale data are due to differences in waste types, moisture content and application rates, buffer, nutrients, etc.; they also reflect the discontinuities to be expected at full-scale installations. Moreover, gas production rates will vary with time as the organic content leached from the refuse in the landfill decreases due to biodegradation and

TABLE 38. SUMMARY OF EXPERIMENTAL OBSERVATIONS OF GAS PRODUCTION FROM MUNICIPAL SOLID WASTE

Reference	Experimental Conditions	Gas Yields, m^3(STP)/kg (dry) Total	CH_4
225	1.2m dia. x 2.3m deep sealed lysimeters; simulated precipitation applied; 7-20°C; pH 5.6 to 5.9; 190-day study	0.006	0.001
180	municipal refuse wetted with digester supernatant	0.013	
181	2.4m dia. x 8.5m underground steel tank; 19-49°C; 900-day study	0.004	
10	carboys filled with 34.5 kg (wet) of mixture of refuse, moisture, sewage sludge buffer; 37°C; 670-day study	0.25	0.13
71	208-liter sealed steel lysimeters; 15-20°C; 300-day study	0.001-0.018	0.001
279, 280	1.8m dia. x 3.7m deep steel lysimeters; simulated annual precipitation/infiltration; 2100-day study	0.003-0.018	
211	3m square x 5.2m deep lysimeters; simulated annual rainfall; shredded refuse; 699-day study	0.007	0.004
29	19-liter lysimeters; shredded waste inoculated with sewage sludge; 410-day study	0.001-0.23	0.001-0.14

washout. In most cases, gas production will remain low for any active landfill area until the first three phases of landfill stabilization depicted in Figure 1 have been completed. Thereafter, gas production rates will increase rapidly to a maximum or peak value during active methanogenic stabilization (Phase IV). For each landfill section, the majority of the methane generated will be released during a relatively short period, i.e., 10 to 20% of the total time required for stabilization, unless restricted by the

TABLE 39. SUMMARY OF EXPERIMENTAL OBSERVATIONS OF GAS PRODUCTION RATES IN SMALL-SCALE LANDFILL SIMULATORS

Reference	Total Gas Production Rate, $m^3/kg \cdot yr$ Average	Maximum
225	0.007	0.007
10	0.13	0.44
71	0.0001-0.013	0.055
211	0.002	0.030
29	0.025-0.488	3.16

factors indicated previously. After the available biodegradable substrate is exhausted, gas production rates will rapidly decline and gas collection for recovery from that landfill would correspondingly become unattractive. Recognition of the sequence of events leading to and controlling high gas (methane) production rates is paramount in planning and designing for efficient and cost effective gas management strategies. These strategies should include consideration of reusable and/or mobile gas collection/recovery appurtenances which could be moved sequentially in a scheduled fashion as the landfill is developed. Such preconceived temporal and spatial planning of gas removal/recovery/utilization facilities within a landfill stabilization perspective has not yet been established as general procedure.

In spite of the previously outlined uncertainties associated with the results of landfill studies, the landfill lysimeter and full-scale data presented in Tables 39 and 40, respectively, tend to correlate fairly well. Simulator studies have generally yielded gas production rates on the order of 0.002-0.13 $m^3/kg \cdot yr$, while full-scale studies have exhibited a range of 0.001-0.008 $m^3/kg \cdot yr$. The higher gas production rates were generally reported for studies using buffer and moisture controls. Therefore, gas production rates of 0.005 to 0.008 m^3/kg dry waste per year may be anticipated from controlled landfills within a few years of refuse placement. However, it should be recognized that higher gas production rates are probably associated with those portions of a landfill that have aged to the active methanogenic phase of landfill stabilization.

Gas Composition

Landfill gases are typically 40 to 60% methane, with the remaining volume comprised primarily of carbon dioxide and 1 or 2% (total) of other miscellaneous inorganic gases and organic vapors. Bench-scale studies with leachate recirculation have achieved methane contents as high as 70%, although methane contents this high have not been common on full-scale. Table 41 provides a summary of gas composition (% CH_4, CO_2, N_2, O_2) for a number of full-scale facilities reviewed by EMCON (1980). Additional data on trace constituents are provided by EMCON (1977) and Lofy (1981) as summarized in Table 42. The data presented indicate that organic and inorganic sulfur

TABLE 40. ON-LINE LANDFILL GAS RECOVERY FACILITIES IN U.S.

	Landfill Characteristics			Gas Recovery Program						
Landfill and Location	Depth, m	Area, $x10^6m^2$	MSW in Place, $x10^6kg$	No. of Gas Wells	Depth of Wells, m	LFG Recovered, $x10^6m^3/day$	Rate of Gas Production $m^3/kg \cdot yr$	LSF Sold to User, $x10^6m^3/day$	Heat Content of Delivered, kJ/m^3	Type of Gas Treatment
Acme, CA	24.3	0.50	2,503.8	12	21.3	0.056	0.008	0.056	13.4	Proprietory
Azusa, CA	51.8	0.30	6,350.3	41	30.4-48.7	0.120	0.002	0.014	13.4	Triethylene glycol
Bradley Sanitary Landfill, CA	30.4-36.5	0.26	8,164.6	39	18.2-23.5	0.076	0.003	0.076	12.1	Dehydration Solids Removal
Cinnaminson, CA	18.3	0.26	2,267.9	29	15.2-18.2	0.019	0.003	0.019	14.7-16.1	Minimal; water separators to remove moisture
City of Industry, CA	39.6	1.21	6,350.3	30	39.6	0.141	0.008	0.07	26.8	Selexol and Proprietory
Davis Street, CA	24.4	0.78	5,252.6	20	18.2	0.084	0.005	0.084	13.4	Proprietory
Industry Hills, CA	9.14-33.5	2.4	3,229.5	30	12.9-22.8	0.006	-	-	13.4	None
North Valley, CA	76.2	0.17	4,535.9	5	30.4	0.031	0.002	0.031	26.8	Triethylene glycol, molecular sieves
Palos Verdes, CA	45.7-76.2	0.17	18,143.7	12	45.7	0.051	0.001	0.021	26.8	Triethylene glycol, molecular sieves
Mountain View, CA	12.2	1.011	3,628	33	13.3	0.085	0.008	0.014	-	Glycol, alumina gel, molecular sieves, activated carbon
Fresh Kills, CA	15.2*	1.61**	68,038.8	123	16.7	0.282	0.001	0.141	26.8	Selexol and
Sheldon-Arleta, Sun Valley, CA	36	0.16	5,450	14	24-33	0.100	0.014	0.10	13.4	Dehydration
Puente Hills, CA	-	-	-	87	36	0.250	-	0.16	13.4	Not available
Monterey Park, CA	75.7	0.50	20,090	56	-	0.220	0.004	0.22	26.8	Chilling, Selexol, other proprietory
Duarte, CA	-	0.13	1,820	33	18	0.030	0.006	-	-	Not available
Scholl Canyon, Glendale, CA	26.0	0.18	4,500	27	25.7-56.1	0.047	0.006	-	-	Proprietory

Compiled from EPA (1979), USDOE (1981), Campbell (1981), and Tour Fact Sheets from the Sixth International GRCDA Landfill Gas Symposium, March (1983).

*Depth of Landfill in the Project area

**Project Area. Total Area = $6.44 \times 10^6 m^2$

TABLE 41. SUMMARY OF LANDFILL GAS COMPOSITION AT FULL-SCALE LANDFILLS

Landfill Site	Gas Composition, % CH_4	CO_2	N_2	O_2
Azuza Western, Azuza, CA	50	50	--	--
Bradley, Los Angeles, CA	50	50	--	--
Central Disposal Site Sonoma Co., CA	50	50	--	--
G.R.O.W.S., Norristown, PA	46	53	1	--
Hewitt, Los Angeles, CA	45	55	--	--
Mountain View, Mountain View, CA	44	34	21	1
Palos Verdes, Rolling Hills, CA	53	43	3	--
P.I.I., Denver, CO	45	55	--	--
Scholl Canyon, Glendale, CA	40	51	7	2
Shelton-Arleta Los Angeles, CA	55	45	--	--

After EMCON, 1980.

TABLE 42. TRACE CONSTITUENTS DETECTED IN LANDFILL GASES

Constituent	EMCON, 1977 (Mountain View) (grains/100scf)*	LOFY, 1981 (Scholl Canyon) (grains/100scf)
Hydrogen Sulfide	0.40-0.91	<0.01
Mercaptan Sulfur	0.0 -0.33	0.01**
Sulfides	0.41-0.80	--
Disulfides and Residuals	0.93-1.65	--
Acetic Acid	--	0.27
Propionic Acid	--	0.41
Butyric Acid	--	0.39
Valeric Acid	--	0.13
Caproic Acid	--	0.08
H_2O Vapor	--	123.0

*$\frac{1 \text{ grain}}{100 \text{ scf}} = 0.0055 \frac{kg}{m^3}$ **Reported as organic sulfur compounds

TABLE 43. REPRESENTATIVE LIST OF ORGANIC COMPOUNDS IDENTIFIED IN LANDFILL GAS

*Pentane
*Dichloromethane
*Hexane
*Iso-octane
*Methylbenzene
*Tetrachloroethene
*Ethylbenzene
*Nonane
*Propylbenzene
Tetramethylhexane
Methylpentane
Dimethylpentane
Methylhexane
Heptane
Trimethylcyclopentane
Dimethylhexane
Dimethylcyclohexane
Octane
Dimethylhexene
Dimethylcyclohexane
Trimethylcyclohexane
Cyclohexyl-eicosane
Ethylpentene
Ethylmethylbutene
Tetramethylpentane
Diethylcyclohexane
Tetramethylbutane
Methylnonene
Tetramethylcyclopentane
Ethylmethylcyclohexane
Methylpropylpentanol
Dichlorofluoromethane
Heptanol
Decane
Decahydronaphthalene
*Dichloroethylene
*Dichloroethane
*Benzene
*Trichloroethylene
*Trichlorethane
*Chlorobenzene
Bimethylbenzene
*Isopropylbenzene
*Napthalene
Methylpentylhydroperoxide
Methylcyclopentane
Hexene
Dimethylcyclopentane
Cycloheptane
Tetrahydrodimethylfuran
Methylheptane
Ethylmethylcyclopentane
Tetramethylcyclopentane
Dimethylheptane
Ethylcyclohexane
Ethylmethylcyclohexane
Methylpropylpentanol
Iso-octanol
Octahydromethylpentalene
Dimethyl(methylpropyl)cyclohexane
Ethylmethylheptane
Methylene-butanediol
Tetramethylhexene
Methylpropylpentanol
Nonyne
Methyl(methylethenyl)-cyclohexene
Hexadiene
Ethylbutanol
Butycyclohexane

After GRI, 1982
*Further quantitative data on these compounds is provided in Table 44.

TABLE 44. SUMMARY OF MAXIMUM AND AVERAGE VOLATILE ORGANIC COMPOUND CONCENTRATIONS (PPM BY VOLUME) FOUND IN FULL-SCALE LANDFILL AND LANDFILL SIMULATOR GASES

Compound	Full-Scale Landfill Gases*				Landfill Simulators**		Regulatory Levels				
	Inlet to Treatment		Product Gas[a]	Surface			TLV[b]	STEL[c]	NYSAAL[f]	OSHA/NIOSH[g]	
	Max	Mean			Max	Mean				PEL/TWA	IDLH
Pentane	5.0	0.4	0.8	0.3	ND	ND	600	750	NR	1000	5000
1,1-dichloroethylene	1.1	0.1	0.2	<0.01	0.55	0.12	5	20	0.02	5	NR
Dichloromethane	12.0	0.9	0.6	0.2	1.57	0.19	100	500	0.33	500	5000
1,2-dichloroethylene	3.6	0.7	0.8	0.2	3.26	0.21	200	250	NR	200	4000
1,1-dichloroethane	7.5	0.4	0.2	0.3	2.08	0.33	200	250	NR	100	4000
Hexane	28.0	1.8	8.3	0.3	97.00	8.83	50	NR[e]	NR	500	5000
Benzene	23.0	1.7	0.7	0.3	12.80	1.64	10	25	0.03	1	2000
Iso-octane	4.1	0.4	0.7	1.0	8.46	0.77	300	375	NR	NR	NR
Trichloroethylene	8.1	0.8	0.7	1.0	1.76	0.37	NR	NR	0.17	100	1000
Methylbenzene	210.0	9.6	2.9	0.3	6.35	1.92	100	150	2.00	200	2000
1,1,2-trichloroethane	0.1	<0.01	<0.01	ND[d]	ND	ND	10	20	0.03	10	500
Tetrachloroethylene	35.0	1.3	0.8	0.3	0.20	0.03	NR	NR	0.17	100	500
Chlorobenzene	11.0	0.4	0.1	ND	0.24	0.04	75	NR	0.33	100	2000
Ethylbenzene	54.0	3.0	1.1	0.2	4.01	0.57	100	125	0.25	75	2400
m,p-xylene	91.0	3.7	1.2	0.8	4.40	0.66	100	150	0.33	100	10000
o-xylene	25.0	1.3	0.4	0.1	2.90	0.50	100	150	0.33	100	10000
Nonane	12.0	0.9	0.7	0.2	6.82	1.28	200	250	NR	200	NR
Isopropylbenzene	28.0	0.7	0.5	<0.01	1.22	0.15	50	75	NR	50	8000
Propylbenzene	3.5	0.1	0.2	<0.01	0.17	0.05	NR	NR	NR	NR	NR
Napthalene	0.1	<0.01	<0.01	ND	0.84	0.14	10	15	0.03	10	500

[a] Including products from a high- and a medium-BTU gas treatment system

[b] Threshhold Limit Value - American Conference of Governmental Industrial Hygenists, Inc., ACGIG, 1982*

[c] Short Term Exposure Limit - ACGIH, 1982*

[d] ND = not detected

[e] NR = not reported

[f] New York State Acceptable Ambient Levels for toxic air contaminants as presented in Air Guide #1, NYSDEC, December 15, 1983.**

[g] Occupational Safety and Health Administration and National Institute for Occupational Safety:

PEL = Permissible Exposure Level averaged over an 8-hour work shift;

IDLH = Maximum Level Immediately Dangerous to Life or Health, i.e., from which one could escape within 30 minutes without irreversible health effects.**

*After GRI (1982)

**After Vogt and Walsh (1984)

compounds may be common trace gaseous constituents and that volatile organic acids were also detected.

Investigations of trace organics in landfill gases have been performed by ESCOR, Inc. for the Gas Research Institute (GRI, 1982). Sixty-nine individual organic compounds were identified by two independent laboratories as summarized in Table 43. Twenty compounds were targeted for further quantitative study and a summary of ESCOR's findings for inlet, processed, and surface gases are compared to the American Conference of Governmental Industrial Hygenists' Threshold Limit Values (TLV) and Short-Term Exposure Limits (STEL), New York State Acceptable Ambient Levels and OSHA/NIOSH limits in Table 44.

Similar studies performed for GRI on gases emanating from landfill simulators containing known quantities of co-disposed industrial waste and priority pollutants have been reported (Vogt and Walsh, 1984). The results of these studies are also presented in Table 44.

COLLECTION AND TREATMENT OF LANDFILL GASES

The equipment required and generally used to collect and treat landfill gases will depend upon the intended use of the gas. Product gases may be withdrawn to prevent migration and simply flared or exhausted to the atmosphere, withdrawn and sold to a consumer directly, used on-site with or without prior treatment, or treated and sold to a consumer as pipeline quality gas.

Landfill Gas Collection--

Gas collection systems employed in practice may consist of simple ventilation and/or flaring systems coupled with shallow trench induced exhaust networks intended primarily for migration control, and/or perforated pipe well matrices placed either vertically or horizonally. The latter are generally used for energy recovery and are reviewed in more detail by Esmaili (1975), Moore and Lynch (1977), Stone (1978), EPA (1979), EMCON (1980) and USDOE (1981).

Induced exhaust well systems are the most popular for energy recovery. These systems will generally encompass extraction equipment such as transport and well piping, backfill gravel, blowers and compressors, metering equipment, and monitoring equipment. Well or trench systems generally incorporate perforated PVC pipe, although polyethylene or fiberglass pipes can also be used. The advantages and disadvantages of these are summarized in Table 45. Networks of header pipes are generally connected to vertical wells which are spaced so that their radii of influence overlap; the radius of influence of wells depend on their depth and the pumping rate (Esmaili, 1975; Moore and Lynch, 1977); Constable, et al., 1979), as well as the degree of compaction, i.e., refuse and cover permeability.

Vertical wells are generally placed to a depth approaching the total refuse depth depending on the existing volume of leachate. The lower half or more of the well piping is usually perforated. Gravel backfill is used for the perforated section, while the upper portion of the boreholes are backfilled with soil to help prevent air intrusion.

TABLE 45. ADVANTAGES AND DISADVANTAGES OF GAS COLLECTION PIPING MATERIALS

PIPING MATERIAL	ADVANTAGES	DISADVANTAGES
Polyvinyl chloride (PVC)	Lightweight, easily installed, corrosion resistant, low cost.	Becomes brittle when exposed to sunlight for extended periods; fails under high differential shear loading.
Polyethylene (PE)	Corrosion resistant; can withstand high bending loads without shear.	Requires special welding equipment for installation; higher cost than PVC.
Fiberglass	Lightweight	High cost; special sealing required to prevent leachate intrusion.
Steel	Can withstand high bending loads without shear.	Subject to corrosion from acids; special welding equipment required; high cost.

After EMCON 1980; Street, 1983; Petro, 1983.

Perforated pipe may also be placed horizonally in a network of shallow trenches, but these must be well sealed at the top to prevent introduction of air. In some cases, shallow gravel-filled trenches have been used without perforated pipe, with the trench serving as the collection system. The success of these systems is highly dependent upon providing an impermeable layer, perhaps a synthetic liner, to prevent air introduction from the surface. The economics of gas collection and liner systems are addressed in detail by EPA (1979).

Centrifugal blowers are often recommended for low vacuum pressures [up to 16 cm (40 in.) water]. These blowers are easily throttled throughout their operating range, although spark-proof varieties are required and are available from several manufactures. For higher pressures, regenerative blowers may be desired. Rotary lobe compressors are generally recommended for landfill gas applications requiring gas pressures in excess of 1425 to 2138 kg/m^2 (2-3 psi).

Gas flow measurement in landfills may be accomplished in gas collection piping using pitot tubes, venturi and orifice plate flow meters, and turbine meters. However, such flow measurements may be difficult to perform accurately and a combination of the above methods, coupled with frequent cross-calibrations of these, is highly recommended.

Landfill Gas Treatment--

As noted previously, the intended use of the gas produced at a particular landfill will dictate the extent of treatment required. Raw landfill gases typically have a low heating value due to the dilution of methane with CO_2, N_2, and possibly O_2. They will likely contain troublesome constituents such as water and hydrogen sulfide. Trace levels of hydrocarbons are also of concern, although these may be expected to oxidize rapidly when the gas is combusted.

Treatment technologies available for the production of either medium BTU (13-15 kJ/m^3; 500-600 BTU/SCF) or pipeline (26 kJ/m^3; 1000+ BTU/SCF) gases are aptly reviewed by EPA (1979), Ashare (1981) and Love (1983). An indication of treatment processes used at currently operating full-scale landfills is presented in Table 40.

Medium BTU Gases--Medium BTU gas is generally produced from raw landfill gas by removing the water vapor and possibly hydrogen sulfide. Condensate and particulates are first removed in a gas/liquid separator; if further water vapor removal is desired, the gas is compressed and cooled prior to being dehydrated using glycol or triethylene glycol. As indicated in Table 46, silica gel, alumina, or molecular sieves may also serve to absorb excess water vapor, although these techniques are generally too expensive for large applications. Glycol absorption is generally the method of choice.

Hydrogen sulfide may be removed using a number of organic solvent absorbents, many of which will absorb CO_2. H_2S can be selectively removed using dry oxidation processes which are also selective for mercaptans, carbon oxysulfide, carbon disulfide and thiophenes. These processes use intermediate oxygen carriers (such as wood shavings) which are nonregenerative and require periodic recharging. This has led to the development of aqueous hydrogen sulfide oxidation methods which utilize solutions or suspensions of sodium carbonates, potassium carbonates, heavy metals (arsenic or iron) or quinones. Continuous operations with recovery of elemental sulfur of high purity are usually possible. However, since these latter processes may be prohibitively expensive for most medium BTU gas applications, solvent methods are generally preferred.

High BTU (Pipeline) Gases--Landfill gas must have a high heat value and a high degree of purity to be sold and mixed with pipeline quality natural gas. Water must be removed to less than 0.0001 kg/m^3 (<7 lbs/MMSCF), hydrogen sulfide to levels ranging from 4 to 80 kg/m^3 or less, and carbon dioxide and nitrogen to sufficiently low levels so that 1000+ BTU/SCF (>26 kJ/m^3) are obtained.

Water can be removed by the previously mentioned silicate absorption processes, or by absorption with glycols or Selexol, a proprietary solvent which also absorbs heavy hydrocarbons. Alternatively, water may be removed by chilling to approximately 35°F (2°C). Heavy hydrocarbons may be removed using absorption with lean oils or ethylene glycol, adsorbed using activated carbon, or by a combination of absorption followed by adsorption.

Carbon dioxide can be removed using aqueous phase organic solvents, alkaline salt solutions or alkanolamines as indicated in Table 46. Solid bed adsorption using activated carbon or molecular sieves (silicates) is also

TABLE 46. SUMMARY OF GAS TREATMENT METHODS AVAILABLE FOR THE REMOVAL OF WATER, HYDROCARBONS, CO_2, and H_2S

TARGET COMPOUND	TREATMENT PROCESS TYPE	TREATMENT PROCESS ALTERNATIVES AVAILABLE
Water	Adsorption	1. Silica gel, 2. Molecular sieves, and 3. Alumina
	Absorption	*1. Ethylene glycol (at low temperature -20°F) 2. Selexol
	Refrigeration	1. Chilling to 35°C
Hydrocarbons	Adsorption	1. Activated carbon
	Absorption	1. Lean oil absorption, 2. Ethylene glycol, and 3. Selexol all at low temperatures (-20°F, -29°C)
	Combination	*1. Refrigeration with Ethylene glycol plus activated carbon adsorption
CO_2 and H_2S	Absorption	1. Organic Solvents Selexol Fluor Rectisol 2. Alkaline Salt Solutions Hot Potassium and inhibited hot potassium (Benefield and Catacarb Processes) 3. Alkanolamines mono,-di-tri-ethanol amines; diglycolamines; *UCARSOL-CR (proprietary chemical)
	Adsorption	1. Molecular Sieves 2. Activated Carbon
	Membrane Separation	1. Hollow Fiber Membrane

*Designates method of choice (after Love, 1983).

possible, although extreme caution is needed to prevent sieve contamination by water, butanes and heavier compounds. Carbon dioxide may be selectively removed by reverse osmosis processes. However, membrane processes require extensive pretreatment of product gases to avoid scaling or fouling of the membrane surface.

Adsorption processes (Table 46) are generally preferred for CO_2 and H_2S removal. Organic solvents can accommodate high acid gas loadings and require relatively low recirculation rates compared with other methods. Each of these methods have their own advantages and disadvantages, as reviewed by EPA (1979) and Love (1983). Selexol also absorbs heavy organics and water, thereby decreasing its overall affinity for CO_2. Moreover, CO_2 is absorbed only at high pressure and low temperature, therefore, refrigeration is required. The same is true for Rectisol, which operates best at -80°F (-63°C). Alkaline salt processes generally require high pressures [142,560 kg/m^2 (200 psig]. In these processes, hot potassium carbonates or sodium carbonates (sometimes coupled with proprietary inhibitors as in the Benefield and Catacarb processes) serve as buffers to react with acid gases.

Alkanolamine absorption methods have a widespread acceptance for CO_2 removal from natural gas; monoethanol (MEA) and diethanolamines (DEA) have also been successfully applied. MEA is corrosive at 19% concentrations, whereas, DEA may be used at solution strengths approaching 35% without undue corrosion. Therefore, DEA, which does not absorb heavy hydrocarbons and, therefore, selectively removes CO_2, is the generally preferred method of CO_2 removal.

Nitrogen may be removed by liquifying the methane fraction of landfill gas by mechanical refrigeration, leaving the other gas fractions to be exhausted. Considerable refrigeration equipment is required for this process and it is usually prohibitively costly. The best practice is to avoid drawing air into the landfill to the greatest extent possible, thereby minimizing the nitrogen content.

Economics--

The economics of implementing the preceding gas collection and treatment alternatives have been reviewed in detail by others (EPA, 1979). In this review, four gas treatment alternatives were considered including dehydration, dehydration plus CO_2 removal, dehydration plus CO_2 and N_2 removal, and dehydration plus CO_2 removal and propane blending. Each alternative was also analyzed at several gas production rates as summarized in Table 47. Scrutiny of these data indicates the relative increased costs associated with N_2 removal and the importance of minimizing the introduction of air during gas extraction from the landfill. Based upon an energy value equivalent to revenue of $1.9/mmkJ (1979 dollars), the probable payback periods associated with each alternative ranged from <3 years (Alternative I) to 10 to 30 years (Alternatives II and IV) and >30 years (Alternative III).

TABLE 47. RELATIVE ECONOMICS OF SEVERAL GAS TREATMENT ALTERNATIVES

Treatment Alternative	Cost Item	Production Rate, std m^3/min		
Alternative I. Dehydration, compression	INPUT	13.74	34.69	69.38
	OUTPUT	13.03	32.85	65.70
	Capital Cost, M$	636	957	1388
	Annual Operating Cost, M$	185	273	387
	Annual Energy Output, 10^9 KJ	116	291	581
	Energy Cost, $/MM KJ	1.6	0.9	0.7
Alternative II. Dehydration and CO_2 removal	INPUT	47.29	94.45	141.60
	OUTPUT	13.74	27.47	42.34
	Capital Cost, M$	1740	2772	3792
	Annual Operating Cost, M$	359	537	702
	Annual Energy Output, 10^9 KJ	231	463	711
	Energy Cost, $/MM KJ	1.6	1.2	1.0
Alternative III. Dehydration plus CO_2 removal and N_2 removal	INPUT	47.29	94.45	141.60
	OUTPUT	11.89	24.64	40.36
	Capital Cost, M$	2612	4038	5450
	Annual Operating Cost, M$	555	807	1051
	Annual Energy Output, 10^9 KJ	203	424	695
	Energy Cost, $/MM KJ	2.7	1.9	1.5
Alternative IV. Dehydration plus CO_2 removal and propane blending	INPUT	47.29	94.45	141.60
	OUTPUT	14.22	28.43	43.70
	Capital Cost, M$	1802	2847	3877
	Annual Operating Cost, M$	463	730	992
	Annual Energy Production, 10^9 KJ	251	503	773
	Energy Cost, $/MM KJ	1.8	1.5	1.3

SECTION 8

LEACHATE AND SOIL INTERACTIONS

GENERAL PERSPECTIVE

As previously noted, one of the primary concerns associated with landfill disposal of municipal and industrial wastes centers on the formation and migration of leachate into the surrounding environment. Presently, the installation of low permeability clay and/or synthetic liners is mandated to deter this migration and its potential deleterious effects. However, many landfills are in existence which have been constructed without the benefit of such liners. Moreover, clay liners are known to be permeable, and recent evidence has shown the same to be true for synthetic liners (Haxo, 1984; Giroud, 1984). Therefore, the purpose of this section is to introduce literature pertinent to the migration of leachates and their subsequent interactions with surrounding native soils and to use it to evaluate the associated implications in relation to soil types present and the necessity and/or effectiveness of available remedial measures. It is not intended here to provide an exhaustive review, but to expose such environmental impacts should leachate (or gas) migration occur.

Basic research on soil/leachate interactions has been ongoing in the U.S. (Roulier, 1977; Fuller, 1977; Copenhauer and Wilkinson, 1979), Canada (Phillips and Nathawani, 1976) and Europe (Sumner, 1978) since the early 1970's. The scope of this research has been extremely broad in nature due to the wide variability in native soil types and leachate characteristics. To provide for a more focused discussion, the review presented here will be limited to soils comprised of mixtures of sand, silt and clays, with the clays consisting of kaolinitic, illitic, and montmorillonitic minerals. Bentonitic clays were not considered, since these are specifically used in slurry wall systems and the substantial amount of information available on the interaction between a variety of toxic chemicals and slurry walls and slurry trenches is beyond the scope of this review. Nevertheless, since bentonite is in reality a special type of montmorillonite, many of the results and conclusions of this section may be extended to include bentonite slurries.

Soils used for experimentation basically fall into two categories; defined mixtures of different proportions of clay minerals and sands, and natural soils which were considered representative of a particular landfill site. While the former offer valuable insight into specific physico-chemical interactive properties of individual materials and mixtures thereof, the latter are more relevant to actual engineering applications. Recognizing that results from a particular site may not extrapolate well to other landfill sites, the use of defined materials may be preferred to provide boundary expectations of the response of different soil types to applied leachates.

Studies on the interactions of soils with leachates can be broadly classified into four topical areas, each focused on the fate or biological removal processes, as associated with heavy metals, pesticides, organics and selected toxic substances. Accordingly, the following discussion is organized to address each of these areas in turn, followed by a review of attempts at modeling leachate and soil interactions, and a summary and synthesis of recommendations for future research.

Heavy Metal Attenuation

A considerable number of studies have been performed to evaluate interactions between heavy metals in leachate and soils. For the most part, emphasis has been placed on the fates of cadmium (Hem, 1972; Jurinak and Sanitillan-Medrano, 1974; Weber and Posselt, 1975; Stevenson, 1976; Gibb and Cartwright, 1976; Fuller, 1977, 1978; Garcia-Miragaya and Page, 1977; Doner, 1978; Fuller, et al., 1981), nickel (Fuller, 1977; Doner, 1978), lead (Santillan-Medrano and Jurinak, 1976; Stevenson, 1976; Zimdahl and Skogerbee, 1977), zinc (Hem, 1972; Fuller, 1977; Fuller, et al., 1981) and copper (Stevenson, 1976; Doner, 1978).

Results from these studies provide substantive evidence that these metals are mobile in natural soils, even in those soils exhibiting low permeability. The relative mobility of these metals has been found to be a function of several factors including pH, soil types, total organic carbon content of soil organic matter, nature and concentration of metal ions, and the aerobicity (or anaerobicity) of the soil. In general, as pH decreases due to acidic conditions imposed by organic acid formation, metals become more mobile (Gibb and Cartwright, 1976; Harkins, 1977; Theis, 1976, 1977; Griffin and Shimp, 1976; Griffin, et al., 1977; Frost and Griffin, 1977; Zimdahl and Skogerbee, 1977). Korte, et al., (1975) reported that upon application of synthetic acidic leachate to typical natural soils, metals were eluted in the following order: Mn, Co, Ni, Zn, Cu, Cr, Pb, Cd. Using neutral leachates, Farquhar (1977) noted that all trace elements studied were adsorbed to some extent, with Zn and Fe being most strongly attenuated, and Ca and Mn being most mobile. Roulier (1977) reported that Cr, Hg, and Ni were extremely mobile in a wide variety of soils. Niebla, et al., (1976) reported Hg to be more mobile in leachate than in water, while Griffin and Shimp (1976, 1978) indicated that Hg in leachates was significantly attenuated by clay materials. Gibb and Cartwright (1976), Griffin and Shimp (1976), and Griffin, et al., (1977) all reported Cr to be particularly mobile at neutral pH values, since the Cr^{+6} form is more mobile than Cr^{+3}. Therefore, acidic (or "younger") leachates show less Cr mobility in electronegative clay soils than do the less acidic leachates produced during and after the active methanogenic phases of stabilization. Niebla, et al., (1976) noted similar observations with respect to Hg attenuation.

The composition of the leachate (conductivity, total iron, total metals, organics) and the composition and nature of the soil (% clays, pore size distribution, permeability) also play a major role in determining metal mobility (Korte, et al., 1975; Fuller, 1977; Fuller, et al., 1976, 1981). Griffin and Shimp (1976, 1978) suggest that the clay content is important due to its cation exchange properties, and emphasize that the cation exchange capacity (CEC) is more important than total particle surface area. Fuller, et al., (1981) support this notion, and correlate the high mobility of Cr and Se to their low potentials for cationic exchange. In this regard, the presence of high levels of salts, iron, and organics (TOC) will enhance the migration of metals due to a more rapid exhaustion of the native CEC. Highly permeable soils will also encourage greater metal migration due to higher mass flows and reduced contact opportunity, resulting in a lower potential for occurrence of clay precipitation reactions.

Microbial activity can influence metal migration by affecting several of the previously mentioned attenuation mechanisms. Many biochemicals synthesized by microorganisms, including amino acids and the simple aliphatic acids, form soluble complexes with metal ions (Stevenson, 1982). Most important is probably the effect of changing pH; first as a result of acidification and subsequent methanogenesis, secondly as a result of competition for adsorption sites and lastly by a restriction of flow due to clogging of soil pores. Further research on relationships between polyvalent cations and the organic components of soil is warranted, since soil organic constituents can form both soluble and insoluble complexes with metal ions.

Pesticide Migration

Pesticide attenuation in landfills arises from two major mechanisms, i.e., microbial degradation and adsorption. Newman and Downing (1958) and Davidson, et al. (1976, 1978, 1980) have studied the problems of pesticide disposal and have concluded that biological degradation represents the major removal mechanism in soils. The degradability of particular pesticides such as atrazine (Cole, 1976; Dao and Lavy, 1978), triazine (Kaiser, et al., 1970) and parathion (Wolfe, et al., 1973; Katan, et al., 1976) as well as combinations of pesticides (Hubbel, et al., 1973) have also been studied. In general, biological degradabilities varied with soil type and pesticide concentration and although a long lag period was typically observed (especially at high concentrations), in almost all cases the pesticide was eventually degraded.

Partial microbial degradation of many pesticides results in the formation of chemically reactive intermediates. These intermediates can potentially combine with the amino- or carbonyl-containing constituents of soil organic matter. The immobilization of chloroanilines (liberated by partial degradation of phenylamide herbicides) by soil organic matter has been reported (Bartha, 1971; Bartha and Pramer, 1970; Hsu and Bartha, 1974). Acid and base hydrolysis resulted in the partial release of chloroanilines bound to soil organic matters. Additionally, the soil-bound chloroanilines were found to be resistant to microbial degradation (Hsu and Bartha, 1974).

The mechanisms for the adsorption of pesticides by soil organic matter include ion exchange, protonation, H-bonding, van der Waal's forces, and coordination through an attached metal ion. An excellent review of these mechanisms has been provided by Stevenson (1982). In addition to these mechanisms, nonpolar molecules are partitioned onto hydrophobic sites on soil organic matter. Adsorption of pesticides onto different soil types (silts, sands and clay) follows Freundlich isotherms. In addition, adsorption sites become saturated at high pesticide concentrations and a uniform wetting front will be absent (Rao, et al., 1979).

In addition to other factors, the mobility of pesticides in the absence of biological activity are related to their solubility. Most pesticides are relatively insoluble in water, although they may be more soluble in acidic and organic-containing leachates than in water. The interrelationships between solubility, biodegradation, and adsorption in soils remain poorly understood and, therefore, are requisite of further study.

Organics

Leachate-derived organics are important not only with regard to their impact as contaminants, but also with respect to their effects on soil structure and its resultant permeability. Early work on this topic, initiated by Grim (1962), indicated that the solubility of clays in acids is dependent upon several parameters including the nature and concentration of the organic acid present, temperature and the duration of the acid/clay contact period. The dissolution of aluminum and other ions was evident even under exposure to relatively weak acids. These results were supported with experiments by Anderson, et al. (1982) where a weak acid (acetic acid), a weak base (aniline), and paint solvent were used. Tests with laboratory columns and field cells (Brown and Anderson, 1980; Anderson, et al., 1982; Brown, et al., 1983) showed an initial decrease in permeability of the soil, followed by a significant increase in permeability accompanied by a change in permeate color. Dissolution of iron and calcium carbonate was suspected in all cases, and "piping", the formation of a noticeable channel in the soil matrix, was observed. Weak acids were shown to be more reactive than weak bases, although weak bases were also responsible for alteration of the soil structure. However, no piping was observed for weak base applications and an aggregated, plate-like structure was noted following contact with weak base. While the results of Anderson, et al. (1982) showed significant changes in permeability following the passing of only two pore volumes, contrasting results have been presented by Lentz, et al. (1984), who observed no change in permeability following passage of six pore volumes of strong acids or bases. Therefore, unanimity of agreement in the published literature is not available and, of more consequence, effects of aqueous mixtures at varying concentrations often are not perceived due to experimental difficulties and/or the lack of true simulation of landfill leachate contact opportunities.

Anderson, et al. (1982) also conducted similar tests with neutral polar organics such as ethylene glycol, acetone, and methanol, and also with neutral nonpolar organics such as xylene and heptane. In all cases, significant changes in soil permeability were noted, often eventually amounting to a two order of magnitude increase in permeability. Ethylene glycol and acetone produced a pronounced initial decrease in permeability, followed by a gradual increase in permeability. At the completion of each test, the soil samples were inspected and structural changes ranging from block-like structures to shrinkage cracks were observed. Re-introduction of water did not result in reversion to the original permeability. Similar results were noted by Foreman and Daniel (1984) and Acar, et al. (1984a,b). However, Acar, et al. (1984b) found that the actual pore size distribution was basically unaltered upon exposure to organics.

The mechanisms at work appear related to the type of clay present, the dielectric constant and dipole moment of the permeant, and the initial degree of soil saturation. Kaolinites showed the greatest resistance to permeability changes. Foreman and Daniel (1984) showed changes in both plasticity and liquid limits when comparing Atterberg limits tests performed with methanol and water. Kaolinitic samples exhibited decreased Atterberg limits, while illitic and montmorillonitic samples showed increased limits. Although it would be expected that liquid limit alterations may stem from changes in interlayer spacings of clay particles, Anderson, et al. (1982) showed no interlayer spacing changes using X-ray diffraction techniques.

Future investigations into the interactions of organic materials, especially in aqueous solutions of leachate-derived organics, seem warranted in order that the interactive effects and mechanisms of permeability alteration can be established with confidence. Additional studies are also needed to address the long-term stability of altered clay structures and, although soils such as used by Anderson, et al. (1982) would generally be accepted as liner materials based on permeability tests using water or calcium sulfate solutions, they may well be rejected when applied to circumstances where soil contact may occur. Data on these issues are only currently becoming available.

Other Toxic Compounds

The majority of studies performed to evaluate leachate and soil interactions have focused on heavy metals, pesticides, and organic solvents. Several studies on the fate of other known toxic compounds such as arsenic, cyanide, and halogenated organics are also available in the literature. Of these, arsenic is apparently relatively immobile in soils, and its adsorption increases with increasing soil concentrations of iron, iron oxides, and aluminum (Fuller, et al., 1980). Johnson and Lancione (1980) have shown that complete immobilization of arsenic by fixation is feasible. In contrast, cyanide is typically very mobile in soils and is apparently more mobile in water than in "typical" leachates (Alesii and Fuller, 1976), thereby indicating potential reactions between cyanide and other leachate components. Microbial attack on cyanides was noted to be very dependent on cyanide concentration, but was considered a potentially useful means of attenuation. Moreover, cyanide was better attenuated at low pH and in the presence of iron oxides and clays of lower electronegativity such as kaolinite or 1:1 lattice clays.

Halogenated organics such as polychlorinated biphenyls (PCB's), polybrominated biphenyls (PBB's), and hexachlorobenzenes (HCB's) are suspected or known carcinogens which are nonpolar and, therefore, of low solubility in water. In column tests using typical soils and leachates, these compounds were found to be relatively immobile; their mobility was further related to the clay content of the soil (Griffin, 1978; Griffin and Chou, 1980). However, in the presence of organic solvents, PCB's and HCB's were shown to be very mobile (Griffin and Chou, 1980). Unfortunately, these compounds are also biologically refractory and tend to persist in soils, thereby presenting a high potential for eventual migration. Adsorption of these compounds onto clays follows linear Freundlich isotherms and increases as the organic content (TOC) and the surface area of the clay increases (Griffin and Chian, 1980). The more chlorinated biphenyls are less mobile than their less chlorinated counterparts.

Analytical Modeling of Leachate/Soil Interactions

A number of authors have developed mathematical models which attempt to describe the movement of single or combinations of contaminants through soil strata. This work has been concentrated in three main areas, i.e., descriptions of general flow through porous media, predictions of contaminant transport, and predictions of contaminant retention (sorptive or other attenuative characteristics of soils).

Ogata (1961) and Elzy, et al. (1974) have concentrated on the problems of vertical and lateral tansmissivity of liquids in soils, while Perrier and Gibson (1982) focused their efforts on percolation and evapotranspiration. These models face uncertainties associated with descriptions of the geologic features (soil types, thicknesses, porosities, permeabilities) of a site which must be incorporated into a quantification of leachate flow. Using finite elements methods, Pinder (1973) and Segol (1977) have attempted to model the potential for leachate contamination of groundwater supplies, as have Pickens and Lennox (1976) and Straub (1980). Sumner (1978) and Pettyjohn, et al. (1981) have focused on the migration of leachate as a plume traversing from beneath the landfill, whereas, several authors have concentrated on dispersion and diffusion processes (Rubin and James, 1973; Van Genuchten, et al. 1977). Other researchers have focused on reactions occurring between the subsurface soil and occluded water (Van Genuchten, et al., 1974; Selim, 1976; Dragun and Helling, 1981).

Some investigators have attempted to describe the fate of specific pollutants such as nickel and cadmium (Fuller, et al., 1981); cadmium (O'Donell, et al., 1977); iron, manganese, and zinc (Farquhar, 1977); salts (Brunotte, et al., 1970); and pesticides (Davidson, et al., 1980b). Intuitively it would seem that a large number of factors would influence the attenuation of these pollutants, e.g., adsorption, liquid throughput, microbial activity and pH, precipitation, and complexation. Moreover, combinations of these factors would make effective modeling very difficult. Nevertheless, these authors also report successful attempts at verifying their models under controlled and defined conditions. While the models developed may serve to evaluate the relative importance of specific parameters or factors regulating leachate transport under these conditions, it is unlikely at this stage that these models can be successfully extended to field applications. Therefore, models need to be developed and verified under field conditions, providing as much quantitative site data on test conditions, geometry of components utilized and detailed results as possible.

Although the state of knowledge concerning the interaction of soils and leachate has been enhanced over the last decade, particularly with respect to attenuation, mobility and alteration of both leachates and the soils they contact, relatively little is known about the actual changes that occur to the soils themselves. Researchers have recently attempted to quantify the effects on soil permeability, but very little is known about specific changes in soil structure or fabric and the long-term stability of these alterations. Future work should be directed toward quantifying the actual test conditions and the changes in the physical properties of the soil as an aid to understanding the role of the numerous parameters that effect interaction. Additionally, these data would be extremely useful to those developing analytical models to simulate such interactive processes.

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APPENDIX A

BIOLOGICAL TREATMENT DATA

TABLE A-1. Bench-Scale Experimental Data for the Activated Sludge Process Relating θ_c to BOD$_5$, COD, and TOC Removal

Reference	θ_c, days	BOD$_5$, mg/l		COD, mg/l		TOC, mg/l		Removal, %			$\frac{BOD_5}{COD}$	$\frac{COD}{TOC}$	Comments
		Influent	Effluent	Influent	Effluent	Influent	Effluent	BOD$_5$	COD	TOC			
19,20,195	1	8000	7800	9200	6700	-	-	2.5	27.2	-	0.87	-	T = 23-25°C
	5	1550	160	2700	830	-	-	89.7	69.2	-	0.57	-	T = 23-25°C
	5	2900	200	6200	430	-	-	93.1	93.1	-	0.47	-	T = 23-25°C
	5	7010	1400	8800	2300	-	-	80.0	73.8	-	0.80	-	T = 23-25°C
288-290	6	13,600	26	19,300	580	6170	-	99.8	97.0	-	0.70	3.1	A; T = 23-25°C
	9	13,600	20	19,300	470	6170	-	99.9	97.6	-	0.70	3.1	A; T = 23-25°C
	20	13,600	6	19,300	300	6170	-	>99.9	98.4	-	0.70	3.1	A; T = 23-25°C
118,119	15	13,600	20	19,300	420	6170	-	99.8	97.8	-	0.70	3.1	A; T = 24°C
	25	13,600	12	19,300	360	6170	-	99.9	98.1	-	0.70	3.1	A; T = 24°C
272	1.5	220	37	-	-	230	151	83.1	-	34.3	-	-	T = 18°C
	3.8	220	20	-	-	230	136	90.9	-	40.1	-	-	T = 18°C
	7.1	220	25	-	-	230	130	88.6	-	43.5	-	-	T = 18°C
53,54,97	2	7100	7100	15,800	15,800	4600	-	0	0	-	0.45	3.4	T = 22°C
	5	7100	3400	15,800	8450	4600	1810	52.1	46.5	60.7	0.45	3.4	T = 22°C
	10	7100	26	15,800	360	4600	140	99.6	97.6	92.6	0.45	3.4	T = 22°C
	10	7100	12	15,800	310	4600	76	99.9	98.0	98.4	0.45	3.4	A; T = 22°C
176,269,270	10	36,000	130	48,000	1550	15,400	-	99.6	96.8	-	0.75	3.1	A; T = 21-25°C
	20	36,000	32	48,000	590	15,400	-	99.9	98.8	-	0.75	3.1	A; T = 21-25°C
	30	36,000	27	48,000	460	15,400	-	99.9	99.0	-	0.75	3.1	A; T = 21-25°C
	30	36,000	90	48,000	610	15,400	-	99.7	98.7	-	0.75	3.1	A; T = 21-25°C
	45	36,000	66	48,000	430	15,400	-	99.8	99.1	-	0.75	3.1	A; T = 21-25°C
	60	36,000	75	48,000	390	15,400	-	99.8	99.2	-	0.75	3.1	A; T = 21-25°C
35,143	5.2	2700	10	3500	90	-	-	99.6	97.4	-	0.77	-	T = 25°C
	3.3	2700	20	3500	100	-	-	99.3	97.1	-	0.77	-	T = 25°C
	3.0	2900	25	3500	110	-	-	99.1	96.9	-	0.83	-	T = 25°C
	5.0	-	-	3800	170	-	-	-	95.5	-	~0.80	-	T = 25°C
	3.0	-	-	3800	1500	-	-	-	60.5	-	~0.80	-	T = 25°C
	2.3	-	-	3800	2200	-	-	-	42.1	-	~0.80	-	T = 25°C
	10.7	~300	-	530	340	135	100	-	35.8	26	~0.66	3.9	T = 15°C
	5.0	~300	-	530	345	135	100	-	34.9	26	~0.66	3.9	T = 15°C
159	10	-	-	1990	1340	602	436	-	32.0	27	-	3.1	
35,143	3.0	~300	-	530	340	135	85	-	35.8	37	~0.66	3.9	T = 15°C
	1.6	~300	-	530	340	135	100	-	35.8	26	~0.66	3.9	T = 15°C
	0.9	~300	-	530	350	135	110	-	34.0	19	~0.66	3.9	T = 15°C
	0.6	~300	-	530	350	135	110	-	34.0	19	~0.66	3.9	T = 15°C
	4.1	~300	-	420	260	130	74	-	38.1	43	~0.66	3.2	T = 13°C
	1.8	~300	-	420	260	130	79	-	38.1	39	~0.66	3.2	T = 13°C
	1.0	~300	-	420	270	130	78	-	35.7	40	~0.66	3.2	T = 13°C
	0.6	~300	-	420	280	130	90	-	33.3	31	~0.66	3.2	T = 13°C
	0.4	~300	-	420	330	130	90	-	21.4	31	~0.66	3.2	T = 13°C
	0.22	~300	-	420	350	130	100	-	16.7	23	~0.66	3.2	T = 13°C
	3.9	~300	-	400	230	105	67	-	42.5	36	~0.66	3.8	T = 16°C
	2.2	~300	-	400	250	105	68	-	37.5	36	~0.66	3.8	T = 16°C
	1.5	~300	-	400	260	105	73	-	35.0	30	~0.66	3.8	T = 16°C
	1.2	~300	-	450	270	140	100	-	40.0	29	~0.66	3.2	T = 16°C
	0.67	~300	-	450	280	140	100	-	37.8	29	~0.66	3.2	T = 16°C
	0.35	~300	-	450	320	140	105	-	28.9	25	~0.66	3.2	T = 16°C
	45	5250	-	9400	150	1700	40	-	98.4	97.6	0.56	5.5	T = 12°C
	20	5250	-	9400	240	1700	77	-	97.4	95.5	0.56	5.5	T = 12°C
	9.9	5250	-	9400	1200	1700	220	-	87.2	87.1	0.56	5.5	T = 12°C
35,143	3.3	~1000	-	1260	220	310	71	-	82.5	77.1	~0.80	4.1	T = 12°C
	1.7	~1000	-	1260	230	310	78	-	81.7	74.8	~0.80	4.1	T = 12°C
	0.[illegible]	~1000	-	1260	600	310	190	-	52.4	39	~0.80	4.1	T = 12°C
	2.0	~680	-	730	430	200	140	-	41.1	30	~0.80	3.7	T = 5°C
	1.9	~680	-	730	340	200	110	-	53.4	45	~0.80	3.7	T = 10°C
	1.9	~680	-	730	300	200	94	-	58.9	53	~0.80	3.7	T = 12°C
	1.8	~680	-	730	260	200	89	-	64.4	56	~0.80	3.7	T = 18°C
	1.9	~680	-	730	230	200	68	-	68.5	66	~0.80	3.7	T = 25°C
151	0.083	-	-	4500	445	1750	400	-	90.1	77.1	-	2.6	A; T = 23°C
	0.25	-	-	4500	565	1750	650	-	87.4	62.9	-	2.6	A; T = 23°C
	0.42	-	-	4500	510	1750	670	-	88.7	61.7	-	2.6	A; T = 23°C
	0.63	-	-	4500	270	1750	300	-	94.0	82.9	-	2.6	A; T = 23°C
	0.[illegible]	-	-	2420	1860	620	500	-	23	19	-	3.9	A; T = 23°C
	0.21	-	-	2600	650	720	440	-	75.0	39	-	3.6	A; T = 23°C
	0.58	-	-	2600	700	770	330	-	73.0	57	-	3.4	A; T = 23°C
	0.21	-	-	1550	850	-	-	-	45	-	-	-	A; T = 23°C
205-207	0.10	260	75	500	290	320	240	71.2	42	25	0.52	1.6	T = 23°C
	0.13	260	42	500	250	320	200	83.8	50	38	0.52	1.6	T = 23°C
	0.23	260	36	500	205	320	140	86.2	59	56	0.52	1.6	T = 23°C
	0.33	260	30	500	210	320	150	88.5	58	53	0.52	1.6	T = 23°C
26	2.5	8300	2000	-	-	-	-	76	-	-	0.5	-	T = 23°C
	5	6400	260	-	-	-	-	96	-	-	0.5	-	T = 23°C
	10	7600	76	-	-	-	-	99	-	-	0.5	-	T = 23°C
	10	6400	770	-	-	-	-	88	-	-	0.5§	-	T = 23°C
	30	7900	240	-	-	-	-	97	-	-	0.5§	-	T = 23°C
228	6	7350	30	9950	240	3550	-	99.6	97.6	-	0.74	2.8	A; T = 18-22°C
	12	7350	30	9950	240	3550	-	99.6	97.6	-	0.74	2.8	A; T = 18-22°C
	20	7350	25	9950	200	3550	-	99.7	98.0	-	0.74	2.8	A; T = 18-22°C
	30	7350	25	9950	200	3550	-	99.7	98.0	-	0.74	2.8	A; T = 18-22°C
	6	7350	3820	9950	6470	3550	-	48	35	-	0.74	2.8	T = 18-22°C
28	10	2220	-	9760	1150	3200	-	-	88.2	-	0.23	3.1	T = 22-23°C
	15	2220	-	9760	860	3200	-	-	91.2	-	0.23	3.1	T = 22-23°C
	20	2220	-	9760	610	3200	-	-	93.7	-	0.23	3.1	T = 22-23°C
	30	2220	-	9760	470	3200	-	-	95.2	-	0.23	3.1	T = 22-23°C
	10	-	-	7760	1250	-	-	-	83.9	-	-	-	T = 22-23°C, Raw
	10	-	-	7760	1160	-	-	-	85.0	-	-	-	T = 22-23°C, Raw with P add.
	10	-	-	10,400	1130	3200	-	-	89.1	-	-	3.3	T = 22-23°C; Lime treated with P add.
	10	-	-	3800	710	-	-	-	81.7	-	-	-	T = 22-23°C; 1:1 dilution
	10	-	-	1500	375	-	-	-	74.8	-	-	-	T = 22-23°C; 1:5 dilution
244	12.5	5170	25	8140	560	-	-	99.5	93.1	-	0.64	-	T = 18-22°C
	12.5	11,250	25	16,700	2500	-	-	99.8	85.0	-	0.67	-	T = 18-22°C
	4.7	750	20	4000	1600	-	-	97.3	60.0	-	0.19	-	T = 18-22°C
	13	6500	60	11,500	3500	-	-	99.1	69.6	-	0.57	-	T = 18-22°C
222	1.2	2845	837	4805	1554	1620	531	70.6	67.6	67.2	0.59	3.0	T = 10°C, COD:P = 100:1
	2.5	2845	261	4805	697	1620	270	90.8	85.5	83.3	0.59	3.0	T = 10°C, COD:P = 100:1
	5.0	2845	52	4805	300	1620	115	98.2	93.8	92.9	0.59	3.0	T = 10°C, COD:P = 100:1
	10.1	2845	16	4805	220	1620	90	99.4	95.4	94.4	0.59	3.0	T = 10°C, COD:P = 100:1
	15.0	2845	9	4805	160	1620	56	99.7	96.7	96.5	0.59	3.0	T = 10°C, COD:P = 100:1
	20.0	2845	12	4805	155	1620	54	99.8	96.8	96.7	0.59	3.0	T = 10°C, COD:P = 100:1

- Data not given
A; Nutrient adjusted; BOD$_5$:N:P=100:5:1
§Author called these AL

TABLE A-2. Bench-Scale Experimental Data for the Activated Sludge Process Relating BOD$_5$ and COD Loading to BOD$_5$ and COD Removal.

	Loading, kgBOD$_5$ or COD/m^3·day		BOD$_5$, mg/l		COD, mg/l		% Removal			$\frac{BOD_5}{COD}$	$\frac{COD}{TOC}$	Comments
Reference	BOD$_5$	COD	Influent	Effluent	Influent	Effluent	BOD$_5$	COD	TOC			
19,20,195	8.0	9.2	8000	7800	9200	6700	2.5	27.2	-	0.87	-	T = 23-25°C
	0.31	0.55	1550	160	2700	830	89.7	69.2	-	0.57	-	T = 23-25°C
	0.58	1.04	2900	200	6200	430	93.1	93.1	-	0.47	-	T = 23-25°C
	1.40	1.75	7010	1400	8800	2300	80.0	73.8	-	0.80	-	T = 23-25°C
288-290	2.27	3.21	13,600	26	19,300	580	99.8	97.0	-	0.70	3.1	A; T = 23-25°C
	1.51	2.14	13,600	20	19,300	470	99.9	97.6	-	0.70	3.1	A; T = 23-25°C
	0.68	0.96	13,600	6	19,300	300	>99.9	98.4	-	0.70	3.1	A; T = 23-25°C
118,119	0.91	1.29	13,600	20	19,300	420	99.8	97.8	-	0.70	3.1	A; T = 24°C
	0.54	0.77	13,600	12	19,300	360	99.9	98.1	-	0.70	3.1	A; T = 24°C
272	0.14	-	220	37	-	-	83.1	-	34.3	-	-	T = 18°C
	0.06	-	220	20	-	-	90.9	-	40.1	-	-	T = 18°C
	0.03	-	220	25	-	-	88.6	-	43.5	-	-	T = 18°C
53,54,97	3.6	7.9	7100	7100	15,800	15,800	0	0	-	0.45	3.4	T = 22°C
	1.4	3.2	7100	3400	15,800	8450	52.1	46.5	60.7	0.45	3.4	T = 22°C
	0.71	1.6	7100	26	15,800	3[illegible]0	99.6	97.6	92.6	0.45	3.4	T = 22°C
	0.71	1.6	7100	12	15,800	310	99.9	98.0	98.4	0.45	3.4	A; T = 22°C
176,269,270	3.6	4.8	36,000	130	48,000	1550	99.6	96.8	-	0.75	3.1	A; T = 21-25°C
	1.8	2.4	36,000	32	48,000	590	99.9	98.8	-	0.75	3.1	A; T = 21-25°C
	1.2	1.6	36,000	27	48,000	460	99.9	99.0	-	0.75	3.1	A; T = 21-25°C
	1.2	1.6	36,000	[illegible]0	48,000	610	99.7	98.7	-	0.75	3.1	A; T = 21-25°C
	0.8	1.1	36,000	66	48,000	430	99.8	99.1	-	0.75	3.1	A; T = 21-25°C
	0.6	0.8	36,000	75	48,000	390	99.8	99.2	-	0.75	3.1	A; T = 21-25°C
35,143	0.52	0.70	2700	10	3500	90	99.6	97.4	-	0.77	-	T = 25°C
	0.82	1.07	2700	20	3500	100	99.3	97.1	-	0.77	-	T = 25°C
	0.97	1.16	2900	25	3500	110	99.1	96.9	-	0.83	-	T = 25°C
	-	0.76	-	-	3800	170	-	95.5	-	~0.80	-	T = 25°C
	-	1.23	-	-	3800	1500	-	60.5	-	~0.80	-	T = 25°C
	-	1.68	-	-	3800	2200	-	42.1	-	~0.80	-	T = 25°C
	-	0.06	~300		530	[illegible]10	-	35.8	26	~0.66	3.9	T = 15°C
	-	0.11	~300	-	530	345	-	34.9	26	~0.66	3.9	T = 15°C
159	-	0.20	-	-	1990	1340	-	32.0			3.3	T = 15°C
35,143	-	0.18	~300	-	530	340	-	35.8	37	~0.66	3.9	T = 15°C
	---	0.32	~300	-	530	340	-	35.8	26	~0.66	3.9	T = 15°C
	---	0.59	~300	-	530	350	-	34.0	19	~0.66	3.9	T = 15°C
	---	0.92	~300	-	530	350	-	34.0	19	~0.66	3.9	T = 15°C
	---	0.10	~300	-	420	260	-	38.1	43	~0.66	3.2	T = 13°C
	--	0.23	~300	-	420	260	-	38.1	39	~0.66	3.2	T = 13°C
	-	0.41	~300	-	420	270	-	35.7	40	~0.66	3.2	T = 13°C
	-	0.70	~300	-	420	280	-	33.3	31	~0.66	3.2	T = 13°C
	-	1.02	~300	-	420	330	-	21.4	31	~0.66	3.2	T = 13°C
	-	1.89	~300	-	420	350	-	16.7	23	~0.66	3.2	T = 13°C
	-	0.10	~300	-	400	230	-	42.5	36	~0.66	3.8	T = 16°C
	-	0.18	~300	-	400	250	-	37.5	36	~0.66	3.8	T = 16°C
	-	0.26	~300	-	400	260	-	35.0	30	~0.66	3.8	T = 16°C
	-	0.37	~300	-	450	270	-	40.0	29	~0.66	3.2	T = 16°C
	-	0.67	~300	-	450	280	-	37.8	29	~0.66	3.2	T = 16°C
	-	1.29	~300	-	450	320	-	28.9	25	~0.66	3.2	T = 16°C
	-	0.21	5250	-	9400	150	-	98.4	97.6	0.56	5.5	T = 12°C
	--	0.46	5250	-	9400	240	-	97.4	95.5	0.56	5.5	T = 12°C
	-	0.95	5250	-	9400	1200	-	87.2	87.1	0.56	5.5	T = 12°C
35,143	-	0.[illegible]	~1000	-	1260	220	-	82.5	77.1	~0.80	4.1	T = 12°C
	-	0.73	~1000	-	1260	230	-	81.7	74.8	~0.80	4.1	T = 12°C
	-	1.44	~1000	-	1260	600	-	52.4	39	~0.80	4.1	T = 12°C
	-	0.37	~680	-	730	430	-	41.1	30	~0.80	3.7	T = 5°C
	-	0.38	~680	-	730	340	-	53.4	45	~0.80	3.7	T = 10°C
	-	0.38	~680	-	730	300	-	58.9	53	~0.80	3.7	T = 12°C
	-	0.40	~680	-	730	260	-	64.4	56	~0.80	3.7	T = 18°C
	-	0.39	~680	-	730	230	-	68.5	66	~0.80	3.7	T = 25°C
151	-	54.0	-	-	4500	445	-	90.1	77.1	-	2.6	A; T = 23°C
	-	18.0	-	-	4500	565	-	87.4	62.9	-	2.6	A; T = 23°C
	-	11.0	-	-	4500	510	-	88.7	61.7	-	2.6	A; T = 23°C
	-	7.2	-	-	4500	270	-	94.0	82.9	-	2.6	A; T = 23°C
	-	29.0	-	-	2420	1860	-	23	19	-	3.9	A; T = 23°C
	-	13.0	-	-	2600	65[illegible]	-	75.0	39	-	3.6	A; T = 23°C
	--	4.5	-	-	2600	7[illegible]	-	73.0	57	-	3.4	A; T = 23°C
	-	7.4	-	-	1550	850	-	45	-	-	-	A; T = 23°C
205-207	2.7	5.2	260	75	500	290	71.2	42	25	0.52	1.6	T = 23°C
	2.1	4.0	260	42	500	250	83.8	50	38	0.52	1.6	T = 23°C
	1.13	2.2	260	36	500	205	86.2	59	56	0.52	1.6	T = 23°C
	0.78	1.5	260	30	500	210	88.5	58	53	0.52	1.6	T = 23°C
26	3.3	-	8300	2000	-	-	76	-	-	0.5	-	T = 23°C
	1.3	-	6400	260	-	-	96	-	-	0.5	-	T = 23°C
	0.76	-	7600	76	-	-	99	-	-	0.5	-	T = 23°C
	0.64	-	6400	770	-	-	88	-	-	0.5§	-	T = 23°C
	0.26	-	7900	240	-	-	97	-	-	0.5§	-	T = 23°C
228	0.31- 0.61	0.41- 0.82	7350	30	9950	240	99.6	97.6	-	0.74	2.8	A; T = 18-22°C
	0.25- 0.37	0.33- 0.50	7350	25	9950	200	99.7	98.0	-	0.74	2.8	A; T = 18-22°C
28	-	0.98	2220	-	9760	1150	-	88.2	-	0.23	3.1	T = 22-23°C
	-	0.65	2220	-	9760	860	-	91.2	-	0.23	3.1	T = 22-23°C
	-	0.49	2220	-	9760	610	-	93.7	-	0.23	3.1	T = 22-23°C
	-	0.33	2220	-	9760	470	-	95.2	-	0.23	3.1	T = 22-23°C
	-	0.78	-	-	7760	1250	-	83.9	-	-	-	T = 22-23°C, Raw
	-	0.78	-	-	7760	1160	-	85.0	-	-	-	T = 22-23°C, Raw with P add.
	-	1.04	-	-	10,400	1130	-	89.1	-	-	3.3	T = 22-23°C; Lime treated with P add.
	-	0.39	-	-	3800	710	-	81.7	-	-	-	T = 22-23°C; 1:1 dilution
	-	0.13	-	-	1500	375	-	74.8	-	-	-	T = 22-23°C; 1:5 dilution
244	0.41	0.65	5170	25	8140	560	99.5	93.1	-	0.64	-	T = 18-22°C
	0.92	1.34	11,250	25	16,700	2500	99.8	85.0	-	§0.67	-	T = 18-22°C
	0.16	0.85	750	20	4000	1600	97.3	60.0	-	0.19	-	T = 18-22°C
	0.50	0.88	6500	60	11,500	3500	99.1	69.6	-	0.57	-	T = 18-22°C
230	-	0.79	-	-	3940	920	-	76		-	-	T = 23°C; control
	-	0.80	-	-	4020	770	-	81		-	-	T = 23°C; lime +
	-	0.75	-	-	3750	510	-	86		-	-	T = 23°C; Na$_2$Co$_3$ Add
	-	0.86	-	-	4280	850	-	78		-	-	T = 23°C; lime Add.
	-	0.76	-	-	3820	850	-	80		-	-	T = 23°C; NaOH Add
												BOD$_5$:N:P
260,261	0.97	-	19,300	80	30400	-	99.6	-		0.63	-	T = 23°C 100:5:1.1
	0.97	-	19,300	55	30400	-	99.7	-		0.63	-	T = 23°C 100:4:1.1
	0.97	-	19,300	36	30400	-	99.8	-		0.63	-	T = 23°C 100:3.2:1.1
	0.97	-	19,300	300	30400	-	99.4	-		0.63	-	T = 23°C 100:4:0.32
	0.97	-	19,300	1430	30400	-	92.6	-		0.63	-	T = 23°C 100:4:0.12
	0.97	-	19,300	560	30400	-	97.1	-		0.63	-	T = 23°C 100:3.2:0.12
291	2.3	3.2	13,600	100	19300	900	99.3	95.3		0.70	3.1	T = 9°C
	2.3	3.2	13,600	100	19300	900	99.3	95.3		0.70	3.1	T = 9°C; A
222	2.4	4.0	2845	837	4805	1554	70.6	67.6		0.59	3.0	T = 10°C; COD:P=100:1
	1.1	1.9	2845	261	4805	697	90.8	85.5		0.59	3.0	T = 10°C; COD:P=100:1
	0.6	0.95	2845	52	4805	300	98.2	93.8		0.59	3.0	T = 10°C; COD:P=100:1
	0.3	0.48	2845	16	4805	220	99.4	95.4		0.59	3.0	T = 10°C; COD:P=100:1
	0.2	0.32	2845	9	4805	160	99.7	96.7		0.59	3.0	T = 10°C; COD:P=100:1
	0.15	0.24	2845	12	4805	155	99.8	96.8		0.59	3.0	T = 10°C; COD:P=100:1

- Data not given

A; Nutrient adjusted; BOD$_5$:N:P=100:5:1

§Author called these N

TABLE A-3. Bench-Scale Experimenta Data for the Activated Process Relating Food to Microorganism (F/M) Ratio to BOD_5 and COD Removal.

Reference	F/M, $\frac{kgBOD_5 \text{ or } COD}{kg\ MLVSS\text{-}day}$		BOD_5, mg/l		COD, mg/l		Removal, %		$\frac{BOD_5}{COD}$	$\frac{COD}{TOC}$	Comments
	BOD_5	COD	Influent	Effluent	Influent	Effluent	BOD	COD			
289,290	0.18	0.25	13,600	6	19,300	300	>99.9	98.4	0.70	3.1	T = 24°C; A
	0.35	0.49	13,600	20	19,300	470	99.9	97.6	0.70	3.1	T = 24°C; A
	0.49	0.68	13,600	28	19,300	580	99.8	97.0	0.70	3.1	T = 24°C; A
118,119	0.18	0.26	13,600	12	19,300	360	99.9	98.1	0.70	3.1	T = 24°C; A
	0.29	0.41	13,600	21	19,300	420	99.8	97.8	0.70	3.1	T = 24°C; A
272	0.16	--	220	25	--	--	88.6	--	--	--	T = 18°C
	0.35	--	220	20	--	--	90.9	--	--	--	T = 18°C
	0.72	--	220	37	--	--	83.1	--	--	--	T = 18°C
53,54,97	0.16	0.35	7,100	26	15,800	360	99.6	97.6	0.45	3.4	T = 22°C
	0.43	0.97	7,100	3400	15,800	8450	52.1	46.5	0.45	3.4	T = 22°C
	0.14	0.31	7,100	10	15,800	310	99.9	98.0	0.45	3.4	T = 22°C; A
193,195	--	0.012	230	--	350	36	--	89.8	0.66	--	T = 23°C; 1:25 dilution
	--	0.016	230	--	370	38	--	89.7	0.62	--	T = 23°C; 1:25 dilution
	--	0.025	230	--	370	44	--	88.1	0.62	--	T = 23°C; 1:25 dilution
	--	0.036	230	--	360	45	--	87.5	0.64	--	T = 23°C; 1:25 dilution
	--	0.050	230	--	350	50	--	85.7	0.66	--	T = 23°C; 1:25 dilution
269,270	0.074	0.099	36,000	75	48,000	390	99.8	99.2	0.75	3.1	T = 23°C; A
	0.067	0.090	36,000	66	48,000	430	99.8	99.1	0.75	3.1	T = 23°C; A
	0.11	0.15	36,000	91	48,000	610	99.7	98.7	0.75	3.1	T = 23°C; A
	0.089	0.12	36,000	27	48,000	460	99.9	99.0	0.75	3.1	T = 23°C; A
	0.12	0.16	36,000	32	48,000	590	99.9	98.8	0.75	3.1	T = 23°C; A
	0.22	0.30	36,000	130	48,000	1550	99.6	96.8	0.75	3.1	T = 23°C; A
159	--	0.27	--	--	1,990	1342	--	32.0	--	3.3	T = 15°C
35,143	--	0.26	2700	10	3500	90	99.6	97.4	0.77	--	T = 25°C
	--	0.37	2700	20	3500	100	99.3	97.1	0.77	--	T = 25°C
	--	0.40	2900	25	3500	110	99.1	96.9	0.83	--	T = 25°C
	--	0.28	--	--	3800	170	--	95.5	0.80	--	T = 25°C
	--	0.49	--	--	3800	1500	--	60.5	0.80	--	T = 25°C
	--	0.70	--	--	3800	2200	--	42.1	0.80	--	T = 25°C
	--	0.033	~300	--	530	340	--	35.8	0.66	3.9	T = 15°C
	--	0.062	~300	--	530	345	--	34.9	0.66	3.9	T = 15°C
	--	0.094	~300	--	530	340	--	35.8	0.66	3.9	T = 15°C
	--	0.14	~300	--	530	340	--	35.8	0.66	3.9	T = 15°C
	--	0.31	~300	--	530	350	--	34.0	0.66	3.9	T = 15°C
	--	0.28	~300	--	530	350	--	34.0	0.66	3.9	T = 15°C
	--	0.088	~300	--	420	260	--	38.1	0.66	3.2	T = 13°C
	--	0.12	~300	--	420	260	--	38.1	0.66	3.2	T = 13°C
	--	0.17	~300	--	420	270	--	35.7	0.66	3.2	T = 13°C
	--	0.26	~300	--	420	280	--	33.3	0.66	3.2	T = 13°C
	--	0.34	~300	--	420	330	--	21.4	0.66	3.2	T = 13°C
	--	1.05	~300	--	420	350	--	16.7	0.66	3.2	T = 13°C
	--	0.049	~300	--	400	230	--	42.5	0.66	3.8	T = 16°C
	--	0.10	~300	--	400	250	--	37.5	0.66	3.8	T = 16°C
	--	0.12	300	--	400	260	--	35.0	0.66	3.8	T = 16°C
	--	0.17	300	--	450	270	--	40.0	0.66	3.2	T = 16°C
	--	0.29	~300	--	450	280	--	37.8	0.66	3.2	T = 16°C
	--	0.43	~300	--	450	320	--	28.9	0.66	3.2	T = 16°C
	--	0.08	5250	--	9400	150	--	98.4	0.56	5.5	T = 12°C
	--	0.16	5250	--	9400	240	--	97.4	0.56	5.5	T = 12°C
	--	0.28	5250	--	9400	1200	--	87.2	0.56	5.5	T = 12°C
35,143	--	0.14	~1000	--	1260	220	--	82.5	0.80	4.1	T = 12°C
	--	0.25	~1000	--	1260	230	--	81.7	0.80	4.1	T = 12°C
	--	0.48	~1000	--	1260	600	--	52.4	0.80	4.1	T = 12°C
	--	0.16	~580	--	730	430	--	41.1	0.80	3.7	T = 5°C
	--	0.19	~580	--	730	340	--	53.4	0.80	3.7	T = 10°C
	--	0.16	~580	--	730	300	--	58.9	0.80	3.7	T = 12°C
	--	0.16	~580	--	730	260	--	64.4	0.80	3.7	T = 18°C
	--	0.17	~580	--	730	230	--	68.5	0.80	3.7	T = 25°C
151	--	3.2	--	--	4500	270	--	94.0	--	2.6	A; T = 23°C
	--	5.7	--	--	4500	510	--	88.7	--	2.6	A; T = 23°C
	--	6.5	--	--	4500	565	--	87.4	--	2.6	A; T = 23°C
	--	2.3	--	--	4500	445	--	90.1	--	2.6	A; T = 23°C
	--	3.1	--	--	2600	700	--	73.0	--	3.4	A; T = 23°C
	--	9.8	--	--	2600	650	--	75.0	--	3.6	A; T = 23°C
	--	2.9	--	--	2420	1060	--	23.1	--	3.9	A; T = 23°C
	--	1.4	--	--	1590	850	--	45.2	--	--	A; T = 23°C
228	0.4-0.6	0.5-0.8	7350	30	9950	240	99.6	97.6	0.74	2.8	A; T = 18-22°C
	0.10-0.18	0.14-0.24	7350	25	9950	200	99.7	98.0	0.74	2.8	A; T = 18-22°C
28	--	0.35	2220	--	9760	470	--	95.2	0.23	3.1	T = 22-23°C
	--	0.41	2220	--	9760	610	--	93.7	0.23	3.1	T = 22-23°C
	--	0.49	2220	--	9760	860	--	91.2	0.23	3.1	T = 22-23°C
	--	0.60	2220	--	9760	1150	--	88.2	0.23	3.1	T = 22-23°C
	--	0.53	--	--	7760	1250	--	83.9	0.23	3.1	T = 22-23°C, raw
	--	0.50	--	--	7760	1160	--	85.0	0.23	3.1	T = 22-23°C, raw + P add.
	--	0.64	--	--	10400	1130	--	89.1	0.23	3.3	T = 22-23°C, lime treated w/P add.
	--	0.47	--	--	3980	710	--	81.7	0.23	3.3	T = 22-23°C 1:1 dilution
244	--	0.44	--	--	1900	375	--	74.8	--	--	T = 22-23°C, 1:5 dilution
	0.064-0.11	0.10-0.16	5170	25	8140	560	99.5	93.1	0.64	--	T = 18-22°C
	0.02	0.11	750	20	4000	1600	97.3	60.0	0.19	--	T = 18-22°C
	0.07	0.12	6500	60	11500	3500	99.1	69.6	0.57	--	T = 18-22°C
222	2.26	3.81	2845	837	4805	1554	70.6	67.6	0.59	3.0	T = 10°C; COD:P=100:1
	1.03	1.75	2845	281	4805	697	90.8	85.5	0.59	3.0	T = 10°C; COD:P=100:1
	0.39	0.66	2845	52	4805	300	98.2	93.8	0.59	3.0	T = 10°C; COD:P=100:1
	0.20	0.34	2845	16	4805	220	99.4	95.4	0.59	3.0	T = 10°C; COD:P=100:1
	0.15	0.25	2845	9	4805	160	99.7	96.7	0.59	3.0	T = 10°C; COD:P=100:1
	0.11	0.19	2845	12	4805	155	99.8	96.8	0.59	3.0	T = 10°C; COD:P=100:1

A = Nutrient adjusted; BOD_5:N:P=100:5:1

-- = Data not given

TABLE A-4. Bench-Scale Experimental Data for the Activated Sludge Process for Heavy Metal Removal.

		Influent Concentration, mg/l							Removal, %							
Reference	pH	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Comments
45	9	--	--	--	2130	--	--	72	--	--	--	99	--	--	>99	Varied θ_c = 7-86 days.
	9	--	--	--	2130	--	--	72	--	--	--	>99	--	--	>99	Considered AL.
	9	--	--	--	2130	--	--	72	--	--	--	99	--	--	>99	
	9	--	--	--	1020	--	--	55	--	--	--	99	--	--	>99	
	9	--	--	--	1020	--	--	55	--	--	--	99	--	--	>99	
	9	--	--	--	1020	--	--	55	--	--	--	99	--	--	>99	
118,119	8	--	0.10	--	1130	0.028	0.11	31	--	75	--	96	82	75	96	
260,261	8.3	0.072	0.37	--	990	0.17	--	50	97	86	--	97	94	--	97	Varied nutrient addition.
	8.6	0.072	0.37	--	990	0.17	--	50	99	91	--	99	96	--	99	
	8.9	0.072	0.37	--	990	0.17	--	50	99	90	--	>99	98	--	>99	
	8.5	0.072	0.37	--	990	0.17	--	50	99	72	--	97	86	--	96	
	8.7	0.072	0.37	--	990	0.17	--	50	97	89	--	99	97	--	99	
	8.7	0.072	0.37	--	990	0.17	--	50	99	91	--	99	91	--	99	
269,270	8.8	0.39	1.9	--	960	1.44	0.65	220	97	93	--	99	80	--	99	Varied θ_c = 10-60 days.
	8.7	0.39	1.9	--	960	1.44	0.65	220	98	97	--	>99	84	--	99	
	8.5	0.39	1.9	--	960	1.44	0.65	220	99	97	--	99	85	--	>99	
	8.8	0.39	1.9	--	960	1.44	0.65	220	97	97	--	99	84	--	>99	
	8.7	0.39	1.9	--	960	1.44	0.65	220	98	98	--	99	88	--	>99	
	8.6	0.39	1.9	--	960	1.44	0.65	220	99	97	--	99	90	--	>99	
288-290	8.3	0.04	0.44	--	1230	--	0.18	39	95	98	--	>99	--	39	>99	Varied θ_c = 6-20 days
	8.3	0.04	0.44	--	1230	--	0.18	39	95	90	--	99	--	67	99	+ T = 9-25°C
	8.3	0.04	0.44	--	1230	--	0.18	39	98	98	--	>99	--	61	>99	
	8.3	0.04	0.44	--	1230	--	0.18	39	85	97	--	99	--	61	99	
	8.3	0.04	0.44	--	1230	--	0.18	39	85	94	--	98	--	56	98	
	8.3	0.04	0.44	--	1230	--	0.18	39	95	98	--	99	--	44	>99	
168	8.4	0.0015	0.017	0.030	20.2	0.045	0.002	1.17	>67	47.1	--	96	94	--		θ_c = 3 days
	8.4	0.0015	0.017	0.030	20.2	0.045			>67	58.8	--	94	94	--		5 days
	8.4	0.0015	0.017	0.030	20.2	0.045			>67	64.7	--	97	94	--		7 days
53,54,97	8.4	--	--	--	240	--	--	--	--	--	--	>96	--	--	--	Varied θ_c = 5-10days.
	8.4	--	--	--	240	--	--	--	--	--	--	>96	--	--	--	
	7.6	--	--	--	240	--	--	--	--	--	--	>96	--	--	--	
	7.6	--	--	--	240	--	--	--	--	--	--	>96	--	--	--	
151	7	--	--	--	290	--	--	--	--	--	--	97	--	--	--	
222	8.6	<0.005	0.14	0.08	102	0.11	0.18	17.6	--	78	>90	98	73	39	95	θ_c = 10 days, T = 10°C

-- Data not given

TABLE A-5. Bench-Scale Experimental Data for the Activated Sludge Process for Alkali and Alkaline Earth Metal Removal.

		Influent Concentration, mg/l					Removal, %					
Reference	pH	Ca	Mg	Mn	K	Na	Ca	Mg	Mn	K	Na	Comments
45	9	3780	660	--	1240	1350	99	79	--	22	35	Varied θ_c = 7-86 days. Considered AL
	9	3780	660	--	1240	1350	99	82	--	23	27	
	9	3780	660	--	1240	1350	99	81	--	32	32	
	9	3010	310	--	500	810	99	74	--	11	20	
	9	3010	310	--	500	810	99	79	--	8	19	
	9	3010	310	--	500	810	99	76	--	17	24	
53,54,97	8.4	1200	170	--	--	--	97	18	--	--	--	Varied θ_c = 5-10 days. Control
	8.4	1200	170	--	--	--	98	29	--	--	--	Lime Addition
	7.6	1200	170	--	--	--	65	26	--	--	--	Lime Addition
	7.6	1200	170	--	--	--	64	18	--	--	--	Control
118,119	8	--	69	13	--	--	--	36	>96	--	--	
151	7	88	100	3.0	900	--	97	90	90	16	--	
206-208	8.2	100	35	--	200	430	66	11	--	30	0	Varied θ_c = 2.3-8 hours.
	8.2	100	35	--	200	430	71	3	--	20	0	
	8.2	100	35	--	200	430	75	14	--	30	0	
	8.2	100	35	--	200	430	75	9	--	20	0	
260,261	8.3	--	--	35	--	--	--	--	98	--	--	Varied nutrient addition.
	8.6	--	--	35	--	--	--	--	97	--	--	
	8.9	--	--	35	--	--	--	--	99	--	--	
	8.5	--	--	35	--	--	--	--	90	--	--	
	8.7	--	--	35	--	--	--	--	97	--	--	
	8.7	--	--	35	--	--	--	--	97	--	--	
168	8.4	550	39.2	4.1	44	120	69	5	99	0	0	θ_c = 3 days
	8.4	550	39.2	4.1	44	120	69	5	97	0	0	5 days
	8.4	550	39.2	4.1	44	120	69	5	97	0	0	7 days
269,270	8.8	1400	310	41	1060	--	99	62	96	35	--	Varied θ_c = 10-60 days.
	8.7	1400	310	41	1060	--	98	71	98	33	--	
	8.5	1400	310	41	1060	--	98	73	99	38	--	
	8.8	1400	310	41	1060	--	95	73	99	36	--	
	8.7	1400	310	41	1060	--	96	64	>99	42	--	
	8.6	1400	310	41	1060	--	94	68	>99	46	--	
288-290	8.3	775	72	14	--	--	94	48	99	--	--	Varied θ_c = 6-20 days + T = 9-25°C.
	8.3	775	72	14	--	--	94	45	99	--	--	
	8.3	775	72	14	--	--	90	58	99	--	--	
	8.3	775	72	14	--	--	95	50	99	--	--	
	8.3	775	72	14	--	--	95	48	99	--	--	
	8.3	775	72	14	--	--	93	60	99	--	--	
222	8.6	348	37	23	153	180	88	11	93	22	0	θ_c = 10 days, T = 10°C

-- Data not given.

TABLE A-6. Bench-Scale Data for Nitrogen Conversion and Removal for the Aerobic Processes.

Reference	θ_c, days	Loading, kgBOD$_5$ or COD/m^3·day		Influent N, mg/l			Effluent N, mg/l			Nitrification, %	TKN Conversion, %	pH	Comments*
		BOD$_5$	COD	TKN	NH$_3$-N	NO$_3$-N	TKN	NH$_3$-N	NO$_3$-N				
35,143	4.1	-	0.10	110	-	3.2	34	-	76	69	69	8.3	T = 13°C; AS
	1.8	-	0.23	110	-	3.2	35	-	75	68	68	8.2	T = 13°C; AS
	1.0	-	0.41	110	-	3.2	29	-	74	67	74	8.0	T = 13°C; AS
	0.6	-	0.70	110	-	3.2	33	-	73	66	70	7.9	T = 13°C; AS
	0.4	-	1.02	110	-	3.2	56	-	48	44	49	8.1	T = 13°C; AS
	0.22	-	1.89	110	-	3.2	93	-	7	6.4	15	8.1	T = 13°C; AS
	3.9	-	0.10	115	-	0.7	34	-	81	70	69	8.2	T = 16°C; AS
	2.2	-	0.18	115	-	0.7	34	-	81	70	69	8.2	T = 16°C; AS
	1.5	-	0.26	115	-	0.7	33	-	80	70	71	8.1	T = 16°C; AS
	1.2	-	0.37	134	134	0.1	36	0.4	85	63	73	8.1	T = 16°C; AS
	0.67	-	0.67	134	134	0.1	32	0.4	87	65	76	8.1	T = 16°C; AS
	0.35	-	1.29	134	134	0.1	118	118	13	10	12	8.2	T = 16°C; AS
	3.3	-	0.38	169	-	0.1	45	-	124	73	73	8.3	T = 12°C; AS
	1.7	-	0.73	176	-	0.1	47	-	129	73	73	8.1	T = 12°C; AS
	0.88	-	1.44	232	-	0.1	232	-	0.2	0	0	8.4	T = 12°C; AS
	2.0	-	0.37	236	228	0.3	228	228	5	2.1	3.4	8.7	T = 5°C; AS
	1.9	-	0.38	239	-	0.3	170	170	72	30	29	8.4	T = 10°C; AS
	1.9	-	0.38	228	-	0.3	55	-	173	76	76	8.0	T = 12°C; AS
	1.8	-	0.40	262	-	0.3	18	-	244	93	93	7.9	T = 18°C; AS
	1.9	-	0.39	268	-	0.3	3	-	265	99	99	7.9	T = 25°C; AS
193,195	6.9	0.033	0.052	12	-	-	5	-	7	58	58	>6.8	T = 23°C; 1:25 dilution; AS
143	τ=37	-	0.011	113	-	3.2	35	-	78	69	69	8.5	T = 12°C; AL
	τ=10	-	0.041	105	-	3.2	65	-	40	38	38	8.4	T = 12°C; AL
	45	-	0.21	250	-	0.1	14	-	6.2	94	94	8.4	T = 12°C; AS
	20	-	0.46	250	-	0.1	29	-	1.6	88	88	8.4	T = 12°C; AS
	9.9	-	0.95	250	-	0.1	102	-	0.2	59	59	8.2	T = 12°C; AS
159	10	-	0.20	-	329	9.7	-	5	334	99	-	6.4	T = 15°C; AS
53,54,97	10	0.71	1.58	280	10	19	16	8.5	4.3		94	8.4	AS; T = 22°C
	10	0.71	1.58	280	10	19	13	6.5	4.1		95	8.4	AS; T = 22°C
	10	0.71	1.58	780	510	19	21	3.4	24		97	7.6	AS; T = 22°C
	10	0.71	1.58	780	510	19	23	4.1	35		97	7.6	AS; T = 22°C
	5	1.42	3.16	280	10	19	118	72	3.9		58	8.0	AS; T = 22°C
269,270	10	3.6	4.8	1390	-	-	29	-	-		98	8.8	AS; T = 23°C
	20	1.8	2.4	1390	-	-	24	-	-		98	8.7	AS; T = 23°C
	30	1.2	1.6	1390	-	-	13	-	-		99	8.5	AS; T = 23°C
	30	1.2	1.6	1770	-	-	70	-	-		96	8.8	AS; T = 23°C
	45	0.8	1.07	1770	-	-	39	-	-		98	8.7	AS; T = 23°C
	60	0.6	0.8	1770	-	-	23	-	-		99	8.6	AS; T = 23°C
244	12.5	-0.65	-1.0	-	970	-	-	8.2	25	-	-	-	T = 18-22°C; AS w/ recycle to denitrification (anaerobic) tank
222	10	0.28	0.48	157	80	-	163	<1	-	-	-	8.6	AS; T = 10°C
	10	0.28	0.48	234	157	-	237	41	-		-	8.6	AS; T = 10°C
	10	0.28	0.48	338	261	-	297	133	-		12	8.6	AS; T = 10°C
	10	0.28	0.48	484	407	-	479	247	-		1	8.6	AS; T = 10°C
	10	0.28	0.48	685	608	-	698	446	-		-	8.6	AS; T = 10°C
	10	0.28	0.48	1051	974	-	986	742	-		6	8.6	AS; T = 10°C
	20	0.14	0.24	161	81	-	<1	<1	-		99	8.6	AS; T = 10°C
	20	0.14	0.24	266	186	-	232	60	-		13	8.6	AS; T = 10°C
	20	0.14	0.24	385	305	-	185	<1	-		52	8.2	AS; T = 10°C
	20	0.14	0.24	541	461	-	195	34	-		64	6.0	AS; T = 10°C
	20	0.14	0.24	778	698	-	452	252	-		42	6.0	AS; T = 10°C
	20	0.14	0.24	1184	1104	-	952	723	-		20	8.3	AS; T = 10°C

- Data not given
* AS = Activated sludge
AL = Aerated lagoon

TABLE A-7. Experimental Data for the Combined Treatment of Leachate with Domestic Wastewater for Bench-Scale and Pilot-Scale Studies Using the Activated Sludge Process.

Reference	Leachate / Domestic WW, %	Combined Quality BOD$_5$, mg/l Influent	Combined Quality BOD$_5$, mg/l Effluent	Combined Quality COD, mg/l Influent	Combined Quality COD, mg/l Effluent	Removal, % BOD$_5$	Removal, % COD	Loading, kg BOD$_5$ or COD/m^3·day BOD$_5$	Loading, kg BOD$_5$ or COD/m^3·day COD	F/M, gBOD$_5$ or COD / gmMLVSS-day BOD$_5$	F/M, gBOD$_5$ or COD / gmMLVSS-day COD	Comments*	BOD$_5$:N:P
19,20	1	225	-	350	24	-	93.1	0.23	0.35	-	-	BS;BOD$_L$=8970;COD$_L$=10,800	
	2	310	-	450	31	-	93.1	0.31	0.45	-	-	BS;BOD$_L$=8970;COD$_L$=10,800	
	5	570	-	770	38	-	95.1	0.57	0.77	-	-	BS;BOD$_L$=8970;COD$_L$=10,800	
	10	1000	-	1300	59	-	95.5	1.00	1.30	-	-	BS;BOD$_L$=8970;COD$_L$=10,800	
	20	1870	-	2360	113	-	95.2	1.87	2.36	-	-	BS;BOD$_L$=8970;COD$_L$=10,800	
44,45,70,73, 286	0.5	270	3	465	35	98.9	92.5	-	-	0.3	0.55	BS;BOD$_L$=24,700;COD$_L$=49,300	
	1	390	3	710	35	99.2	95.1	-	-	0.3	0.55	BD;BOD$_L$=24,700;COD$_L$=49,300	
	2	670	3	1200	40	99.6	96.7	-	-	0.3	0.55	BS;BOD$_L$=24,700;COD$_L$=49,300	
	3	900	3	1690	45	99.7	97.3	-	-	0.3	0.55	BS;BOD$_L$=24,700;COD$_L$=49,300	
	4	1100	5	2160	60	99.5	97.2	-	-	0.3	0.55	BS;BOD$_L$=24,700;COD$_L$=49,300	
	2	670	3	1200	40	99.6	96.7	-	-	0.3	0.55	BS;BOD$_L$=24,700;COD$_L$=49,300	
	2	610	3	-	-	99.5	-	-	-	0.6	1.1	BS;BOD$_L$=24,700;COD$_L$=49,300	
	2	570	6	-	-	98.9	-	-	-	1.0	1.8	BS;BOD$_L$=24,700;COD$_L$=49,300	
42,43	0.5	150	4	250	30	97.3	88.0	-	-	-	-	BS	
59	0.5	-	-	770	135	-	82.5	-	0.51	-	0.51	BS	
	1.0	-	-	870	85	-	90.2	-	0.58	-	0.50	BS	
	2.0	-	-	1070	200	-	81.3	-	0.71	-	0.49	BS	
176,260,261	1	235	5	-	-	97.9	-	0.012	-	-	-	BS;BOD$_L$=19,300	100:8:1
	3	590	14	-	-	97.6	-	0.03	-	-	-	BS;BOD$_L$=19,300	100:5:0.5
	6	1130	11	-	-	99.0	-	0.06	-	-	-	BS;BOD$_L$=19,300	100:4:0.26
	10	1850	8	-	-	99.6	-	0.09	-	-	-	BS;BOD$_L$=19,300	100:3.6:0.3
	20	3640	16	-	-	99.6	-	0.18	-	-	-	BS;BOD$_L$=19,300	100:3.6:0.1
228	2	210	8	335	38	96.3	89.3	0.20	0.25	0.10	0.12	PS; T = 10-15°C	
	3	370	9	550	40	97.5	92.0	0.42	0.57	0.22	0.30	PS; T = 10-15°C	
	8	200	13	380	39	93.6	89.8	0.36	0.46	0.14	0.18	PS; T = 10-15°C	

- Data not given
*BS - Bench-Scale
PS = Pilot-Scale
Subscript L refers to raw leachate concentration in mg/l. All tested at T = 23°C unless specified otherwise.

TABLE A-8. Bench-Scale Experimental Data for the Aerated Lagoon Process.

Reference	τ,days	Organic Loading, kg/m^3·day BOD$_5$	Organic Loading, kg/m^3·day COD	F/M, kg/kgMLVSS·day BOD$_5$	F/M, kg/kgMLVSS·day COD	BOD$_5$, mg/l Inf.	BOD$_5$, mg/l Eff.	COD, mg/l Inf.	COD, mg/l Eff.	TOC, mg/l Inf.	TOC, mg/l Eff.	% Removal BOD$_5$	% Removal COD	% Removal TOC	BOD$_5$/COD	COD/TOC	Comments
26	10	0.64	-	-	-	6400	770	-	-	-	-	88	-	-	0.5	-	T=23°C
	30	0.26	-	-	-	7900	240	-	-	-	-	97	-	-	0.5	-	T=23°C
40-46	7	-	5.0	-	0.37	-	-	35,000	1030	11,800	380	-	97.1	96.8	-	3.0	A*;T=23°C
	15	-	2.3	-	0.20	-	-	35,000	820	11,800	310	-	97.7	97.4	-	3.0	A*;T=23°C
	30	-	1.2	-	0.12	-	-	35,000	540	11,800	210	-	98.5	98.2	-	3.0	A*;T=23°C
	30	-	1.9	-	0.19	-	-	58,000	670	19,400	240	-	98.8	98.8	-	3.0	A*;T=23°C
	60	-	0.97	-	0.11	-	-	58,000	540	19,400	180	-	99.1	99.1	-	3.0	A*;T=23°C
	86	-	0.67	-	0.084	-	-	58,000	415	19,400	160	-	99.3	99.2	-	3.0	A*;T=23°C
35,143	10	-	0.042	-	-	-	-	420	310	132	94	-	26	29	0.66	3.2	T=12°C
	37	-	0.011	-	-	-	-	420	290	132	91	-	31	31	0.66	3.2	T=12°C
244	70	0.01	-	-	-	-	10	-	-	-	-	99	-	-	0.1	-	T=20°C
	70	0.01	-	-	-	-	10	-	-	-	-	99	-	-	0.1-0.4	-	T=20°C
	70	0.02	-	-	-	-	15	-	-	-	-	99	-	-	0.1-0.4	-	T=20°C
	70	0.10-0.40	-	-	-	-	10	-	-	-	-	99	-	-	0.4	-	T=20°C
		0.50	-	-	-	-	10	-	-	-	-	99	-	-	0.4	-	T=20°C
		0.60	-	-	-	-	10	-	-	-	-	99	-	-	0.4	-	T=20°C
		0.70	-	-	-	-	20	-	-	-	-	99	-	-	0.4	-	T=20°C
		0.80	-	-	-	-	60	-	-	-	-	99	-	-	0.4	-	T=20°C
		0.90	-	-	-	-	100	-	-	-	-	98	-	-	0.4	-	T=20°C
	10	1.0	-	-	-	-	200	-	-	-	-	96	-	-	0.4	-	T=20°C
	100	0.09	0.17	-	-	9840	5	17,100	350	-	-	99.8	98.0	-	0.56	-	T=20°C
	82	0.12	0.21	-	-	9840	5	17,100	400	-	-	99.9	97.7	-	0.56	-	T=20°C
	55	0.18	0.31	-	-	9840	5	17,100	350	-	-	99.9	98.0	-	0.56	-	T=20°C
	41	0.24	0.42	-	-	9840	5	17,100	400	-	-	99.9	97.7	-	0.56	-	T=20°C
	100	0.09	0.17	-	-	9840	70	17,100	1200	-	-	99.3	93.0	-	0.56	-	T=5°C
	82	0.12	0.21	-	-	9840	50	17,100	1400	-	-	99.5	91.8	-	0.56	-	T=5°C
	55	0.18	0.31	-	-	9840	40	17,100	1200	-	-	99.6	93.0	-	0.56	-	T=5°C
	41	0.24	0.42	-	-	9840	55	17,100	1200	-	-	99.4	93.0	-	0.56	-	T=5°C
168	2	0.47	0.79	-	0.93	940	28	1,580	275	-	-	97.0	82.6	-	0.6	-	T=21-24°C
	3	0.35	0.59	-	0.61	1040	9	1,760	215	-	-	99.1	87,8	-	0.6	-	T=21-24°C
	5	0.21	0.35	-	0.37	1040	10	1,760	200	-	-	99.0	88.6	-	0.6	-	T=21-24°C
	7	0.15	0.25	-	0.31	1040	7	1,760	175	-	-	99.3	90.1	-	0.6	-	T=21-24°C
	7	0.13	0.22	-	0.33	940	8	1,580	170	-	-	99.1	89.2	-	0.6	-	T=21-24°C
	10	0.09	0.16	-	0.26	940	8	1,580	170	-	-	99.1	89.2	-	0.6	-	T=21-24°C

- Data not given
* Nutrient Adjusted COD:N:P=164:8:1

TABLE A-9. Bench-Scale Experimental Data for the Anaerobic Process Relating θ_c to BOD_5, COD, and TOC Removal.

Reference	Process	θ_c, days	BOD_5, mg/l		COD, mg/l		TOC, mg/l		Removal, %			$\frac{BOD_5}{COD}$	$\frac{COD}{TOC}$	Comments
			Influent	Effluent	Influent	Effluent	Influent	Effluent	BOD_5	COD	TOC			
19,20,196	PFR-SG	5	9100	655	11,200	970	-	-	92.8	91.3	-	0.81	-	T = 23-30°C
		5	1820	165	2240	225	-	-	90.9	90.0	-	0.81	-	T = 23-30°C
		10	18,200	1680	22,400	1780	-	-	90.8	92.1	-	0.81	-	T = 23-30°C
		10	3640	220	4480	380	-	-	93.9	91.5	-	0.81	-	T = 23-30°C
		10	8400	150	10,600	700	-	-	98.2	93.4	-	0.79	-	T = 23-30°C
		12.5	8400	95	10,600	560	-	-	98.9	94.7	-	0.79	-	T = 23-30°C
		12.5	9100	195	11,200	450	-	-	97.9	96.0	-	0.81	-	T = 23-30°C
		20	18,200	790	22,400	1540	-	-	95.7	93.1	-	0.81	-	T = 23-30°C
		20	7300	225	8960	630	-	-	96.9	93.0	-	0.81	-	T = 23-30°C
33,217	CSTR-SG	5	13,000	2150	26,000	8250	9100	-	83.3	68.3	-	0.5	2.9	A; T = 34°C
		10	13,000	935	26,000	7300	9100	-	92.8	71.9	-	0.5	2.9	A; T = 34°C
		20	13,000	435	26,000	4900	9100	-	96.6	81.2	-	0.5	2.9	A; T = 34°C
98,220	CSTR-SG	10	-	-	12,900	1060	4600	280	-	91.8	93.9	~0.45	2.8	A; T = 35°C
		20	-	-	12,900	600	4600	260	-	95.3	94.3	~0.45	2.8	A; T = 35°C
		20	-	-	12,900	630	4600	180	-	95.0	96.1	~0.45	2.8	T = 35°C
		20	-	-	12,900	2860	4600	730	-	77.4	84.1	~0.45	2.8	A; T = 20°C
		20	-	-	12,900	840	4600	230		93.4	95.0	~0.45	2.8	T = 35°C
98,220	PFR-AG	10	-	-	12,900	500	4600	200		96.1	95.7	~0.45	2.8	T = 35°C
143	PFR-AG	1.5	2700	1500	3600	1900	-	-	44.4	47.2	-	0.75	-	T = 25°C
		2.25	2700	2100	3600	3100	-	-	22.2	13.9	-	0.75	-	T = 11°C
		4.5	2700	2050	3600	3000	-	-	24.1	16.9	-	0.75	-	T = 11°C
143	PFR-AG	26	24,500	24,000	38,800	38,800	-	-	2.0	0	-	0.63	-	T = 23°C; 1:11 recycle
		26	24,500	21,800	38,800	35,900	-	-	11.0	7.5	-	0.63	-	T = 23°C; 1:11 recycle
		73	24,500	180	38,800	600	-	-	99.3	98.5	-	0.63	-	T = 23°C; 1:11 recycle
		5.7	5950	110	9100	520	-	-	98.2	94.3	-	0.65	-	T = 23°C; 1:11 recycle
		8.3	5950	50	9100	270	-	-	99.2	97.0	-	0.65	-	T = 23°C; 1:11 recycle
		12	5950	-	9100	720	-	-	-	92.1	-	0.65	-	T = 23°C; 1:11 recycle
		12	5950	65	9100	775	-	-	98.9	91.5	-	0.65	-	T = 23°C; 1:5 recycle
		15	5950	40	9100	420	-	-	99.3	95.4	-	0.65	-	T = 23°C; 1:5 recycle
		18	5950	30	9100	420	-	-	99.5	95.4	-	0.65	-	T = 23°C; 1:5 recycle
		28	5950	55	9100	600	-	-	99.2	93.4	-	0.65	-	T = 23°C; 1:5 recycle
151	CSTR-SG	0.10	3880	3500	6200	6200	2260	2100	10	0	7	0.63	2.7	A; T = 37°C Batch
		0.17	4200	3620	6300	6250	2260	2050	14	<1	9	0.67	2.8	A; T = 37°C Batch
		0.33	4800	4600	6690	5850	2280	2025	4	13	11	0.72	2.9	A; T = 37°C Batch
		1	3600	2800	6120	4760	-	1760	22.2	22.2	-	0.59	~2.6	A; T = 37°C Batch
		5	3400	700	5300	2100	2230	900	79.4	60.4	59.6	0.64	2.4	A; T = 37°C Batch
		10	1300	170	2740	720	830	190	86.9	73.7	77.1	0.47	3.3	A; T = 37°C Batch
		10	1900	80	2900	400	1100	160	95.8	86.2	85.5	0.66	2.6	A; T = 37°C Continuous
		10	1940	135	2480	375	880	190	93.0	84.9	78.4	0.78	2.8	A; T = 37°C Continuous
		15	1530	100	2470	200	810	220	93.5	91.9	72.8	0.62	3.0	A; T = 37°C Continuous
205-207	CSTR-SG	0.1	3700	3400	6000	6000	-	-	8	0	-	0.62	-	A; T = 37°C
		0.16	3700	3400	6000	6000	-	-	8	0	-	0.62	-	A; T = 37°C
		0.33	3700	4100	6000	5400	-	-	0	10	-	0.62	-	A; T = 37°C
		1	3700	2600	6000	4020	-	-	30	33	-	0.62	-	A; T = 37°C
		5	3700	470	6000	1090	-	-	87.3	81.8	-	0.62	-	A; T = 37°C
		10	3700	80	6000	670	-	-	97.8	88.8	-	0.62	-	A; T = 37°C
		15	3700	75	6000	140	-	-	97.8	97.7	-	0.62	-	A; T = 37°C
14	PFR-SG	6.4	-	-	25,000	2000	-	-	-	92	-	-	>3.0	T = 33°C
		6.6	-	-	32,000	8000	-	-	-	75	-	-	>3.0	T = 20°C
44-47	CSTR-AG	7.5	-	-	32,000	2000	-	-	-	93.8	-	-	3.5	T = 23°C; 1:4.4 recycle
		17.5	-	-	32,000	1400	-	-	-	95.6	-	-	3.5	T = 23°C; 1:8.7 recycle
		74	-	-	32,000	1000	-	-	-	96.9	-	-	3.5	T = 23°C; 1:35 recycle
135,136	CSTR-SG	10	3940	320	7350	700	1260	-	91.8	90.5	-	0.54	5.8	T = 35°C
		15	3940	250	7350	650	1260	-	93.6	91.1	-	0.54	5.8	T = 35°C
		20	3940	205	7350	420	1260	-	94.8	94.3	-	0.54	5.8	T = 35°C
223	PFR-AG	1.8	2600	-	3200	1160	-	-	-	63.8	-	0.81	-	T = 24°C
30	CSTR-SG	30	1000	100	-	-	-	-	85	-	-	-	-	T = 24°C
		30	4000	250	-	-	-	-	95	-	-	-	-	T = 24°C
		30	10,000	2000	-	-	-	-	80	-	-	-	-	T = 24°C
237	PFR-AG	0.25	950	-	1870	950	-	-	-	50	-	0.51	2.6	T = 25°C
		0.50	950	-	1870	190	-	-	-	88	-	0.51	2.6	T = 25°C
		1.0	950	-	1870	170	-	-	-	90	-	0.51	2.6	T = 25°C
		2.0	950	-	1870	180	-	-	-	90	-	0.51	2.6	T = 25°C
		3.0	950	-	1870	160	-	-	-	91	-	0.51	2.6	T = 25°C
		2.0	950	-	1870	450	-	-	-	76	-	0.51	2.6	T = 10°C
		3.0	950	-	1870	220	-	-	-	88	-	0.51	2.6	T = 10°C

- Data not given

A = Nutrient Adjusted, BOD_5:N:P = 100:5:1

TABLE A-10. Bench-Scale Experimental Data for the Anaerobic Process Relating BOD_5 and COD Loading to BOD_5 and COD Removal.

		Loading, kg BOD_5 or COD/ $m^3 \cdot day$		BOD_5, mg/l		COD, mg/l		Removal, %				
Reference	Process	BOD_5	COD	Influent	Effluent	Influent	Effluent	BOD_5	COD	$\frac{BOD_5}{COD}$	$\frac{COD}{TOC}$	Comments
19,20,195	PFR-SG	0.36	0.45	1820	165	2240	225	90.9	90.0	0.81	-	T = 23-30°C
		0.36	0.45	3640	220	4480	380	93.9	91.5	0.81	-	T = 23-30°C
		0.37	0.45	7300	225	8960	630	96.9	93.0	0.81	-	T = 23-30°C
		0.67	0.85	8400	95	10,600	560	98.9	94.7	0.79	-	T = 23-30°C
		0.84	1.06	8400	150	10,600	700	98.2	93.4	0.79	-	T = 23-30°C
		0.73	0.90	9100	195	11,200	450	97.9	96.0	0.81	-	T = 23-30°C
		0.91	1.12	18,200	790	22,400	1540	95.7	93.1	0.81	-	T = 23-30°C
		1.82	2.24	9100	655	11,200	970	92.8	91.4	0.81	-	T = 23-30°C
		1.82	2.24	18,200	1680	22,400	1780	90.8	92.1	0.81	-	T = 23-30°C
38,217	CSTR-SG	0.65	1.30	13,000	435	26,000	4900	96.[illegible]	81.2	0.5	2.9	A; T = 34°C
		1.30	2.60	13,000	935	26,000	7300	92.[illegible]	71.9	0.5	2.9	A; T = 34°C
		2.60	5.20	13,000	2150	26,000	8250	83.[illegible]	68.3	0.5	2.9	A; T = 34°C
98,220	CSTR-SG	-	0.65	-	-	12,900	630	-	95.1	~0.45	2.8	T = 35°C
		-	0.65	-	-	12,900	600	-	95.3	~0.45	2.8	T = 35°C; A
		-	1.29	-	-	12,900	1060	-	91.8	-0.45	2.8	T = 35°C; A
		-	0.65	-	-	12,900	2860	-	77.4	~0.45	2.8	T = 20°C; A
		-	0.65	-	-	12,900	840	-	93.5	-0.45	2.8	T = 35°C; lime treated
98,220	PFR-AG	-	1.29	-	-	12,900	500	-	[illegible].1	-0.45	2.8	T = 35°C; lime treated
143	PFR-AG	1.8	2.4	2700	1500	3600	1900	44.4	47.2	0.75	-	T = 25°C
		0.6	0.8	2700	2050	3600	3000	24.1	16.7	0.75	-	T = 11°C
		1.2	1.6	2700	2100	3600	3100	22.2	13.9	0.75	-	T = 11°C
143	PFR-AG	0.34	0.53	24,500	180	38,800	600	99.3	98.5	0.63	-	T = 23°C
		0.94	1.5	24,500	24,000	38,800	38,800	2.0	0	0.63	-	T = 23°C
		0.94	1.5	24,500	21,800	38,800	35,900	11.0	7.5	0.63	-	T = 23°C; 1:11 recycle
		0.21	0.33	5950	55	9100	600	99.2	93.4	0.65	-	T = 23°C; 1:11 recycle
		0.40	0.61	5950	40	9100	420	99.3	95.4	0.65	-	T = 23°C; 1:11 recycle
		0.50	0.76	5950	65	9100	780	98.9	91.5	0.65	-	T = 23°C; 1:5 recycle
		0.72	1.1	5950	50	9100	270	99.2	97.0	0.65	-	T = 23°C; 1:5 recycle
		0.50	0.76	5950	-	9100	720	-	92.1	0.65	-	T = 23°C; 1:5 recycle
		0.33	0.51	5950	30	9100	420	99.5	95.4	0.65	-	T = 23°C; 1:5 recycle
		1.04	1.6	5950	110	9100	520	98.2	94.3	0.65	-	T = 23°C; 1:5 recycle
151	CSTR-SG	0.10	0.17	1530	100	2470	200	93.5	91.9	0.62	3.0	T = 37°C; A; Batch
		0.19	0.25	1940	135	2480	375	93.0	84.9	0.78	2.8	T = 37°C; A; Batch
		0.19	0.29	1900	80	2900	400	95.8	86.2	0.66	2.6	T = 37°C; A; Batch
		0.13	0.27	1300	170	2740	720	86.9	73.7	0.47	3.3	T = 37°C; A; Batch
		0.68	1.06	3400	700	5300	2100	79.4	60.4	0.64	2.4	T = 37°C; A; Batch
		3.6	6.1	3600	2800	6120	4760	22.2	22.2	0.59	~2.6	T = 37°C; A; Batch
		14	20	4800	4600	6690	5850	4	13	0.72	2.9	T = 37°C; A; Continuous
		25	39	4200	3620	6300	6250	14	<1	0.67	2.8	T = 37°C; A; Continuous
		37	60	3880	3500	6200	6200	10	0	0.63	2.7	T = 37°C; A; Continuous
205-207	CSTR-SG	0.25	0.40	3700	75	6000	140	97.[illegible]	97.7	0.62	-	T = 37°C; A
		0.37	0.60	3700	80	6000	670	97.[illegible]	88.8	0.62	-	T = 37°C; A
		0.74	1.2	3700	470	6000	1090	87.3	81.8	0.62	-	T = 37°C; A
		3.7	6	3700	2600	6000	4020	30	33	0.62	-	T = 37°C; A
		11.2	18	3700	4100	6000	5400	0	10	0.62	-	T = 37°C; A
		23	38	3700	3400	6000	6000	8	0	0.62	-	T = 37°C; A
		37	60	3700	3400	6000	6000	8	0	0.62	-	T = 37°C; A
14	PFR-SG	-	4.0	-	-	25,000	2000	-	92	-	>3.0	T = 33°C
		-	4.9	-	-	32,000	8000	-	75	-	>3.0	T = 20°C
44-47	CSTR-AG	-	0.89	-	-	32,000	1000	-	96.9	-	3.5	T = 23°C; 1:35 recycle
		-	2.6	-	-	32,000	1400	-	95.6	-	3.5	T = 23°C; 1:8.7 recycle
		-	5.3	-	-	32,000	2000	-	93.8	-	3.5	T = 23°C; 1:4.4 recycle
135-136	CSTR-SG	0.20	0.37	3940	205	7350	420	94.8	94.3	0.54	5.8	T = 35°C
		0.26	0.49	3940	250	7350	650	93.6	91.1	0.54	5.8	T = 35°C
		0.39	0.74	3940	320	7350	700	91.8	90.5	0.54	5.8	T = 35°C
223	PFR-AG	1.4	1.8	2600	-	3200	1160	-	63.8	0.81	-	T = 24°C; lime treated
30	CSTR-SG	0.03	-	1000	100	-	-	85	-	-	-	T = 24°C
		0.13	-	4000	250	-	-	95	-	-	-	T = 24°C
		0.33	-	10000	2000	-	-	80	-	-	-	T = 24°C
237	PFR-AG	3.8	7.6	950	-	1870	950	-	50	-	-	T = 25°C
		1.9	3.8	950	-	-	190	-	88	-	-	T = 25°C
		1.0	2.0	950	-	-	170	-	90	-	-	T = 25°C
		0.5	1.0	950	-	-	180	-	90	-	-	T = 25°C
		0.3	0.64	950	-	-	160	-	91	-	-	T = 10°C
		0.5	1.0	950	-	-	450	-	76	-	-	T = 10°C
		0.3	0.64	950	-	-	220	-	88	-	-	T = 10°C

- Data not given
A = Nutrient Adjusted; BOD_5:N:P = 100:5:1

TABLE A-11. Bench-Scale Experimental Data for the Anaerobic Process for Methane and Gas Production.

Reference	Process	Influent, mg/l		Loading, kg BOD$_5$ or COD/m^3·day		θ_c, days	CH_4 Production, l/kg BOD$_5$ or COD destroyed		Gas Production, l/kg BOD$_5$ or COD destroyed		Gas Composition, %			$\frac{BOD_5}{COD}$	$\frac{COD}{TOC}$	Comments
		BOD$_5$	COD	BOD$_5$	COD		BOD$_5$	COD	BOD$_5$	COD	CH_4	CO_2	N_2			
19,20,195	PFR-SG	8400	10,600	0.84	1.06	10	-	-	545	454	-	-	-	0.79	-	T = 23-30°C
		8400	10,600	0.67	0.85	12.5	-	-	487	403	-	-	-	0.79	-	T = 23-30°C
		18,200	22,400	1.82	2.24	10	-	-	384	308	-	-	-	0.81	-	T = 23-30°C
		9100	11,200	1.82	2.24	5	-	-	412	340	-	-	-	0.81	-	T = 23-30°C
		18,200	22,400	0.91	1.12	20	-	-	398	332	-	-	-	0.81	-	T = 23-30°C
		9100	11,200	0.73	0.90	12.5	-	-	418	346	-	-	-	0.81	-	T = 23-30°C
		7300	8960	0.37	0.45	20	-	-	346	294	-	-	-	0.81	-	T = 23-30°C
		3640	4480	0.36	0.45	10	-	-	398	332	-	-	-	0.81	-	T = 23-30°C
		1820	2240	0.36	0.45	5	-	-	213	175	-	-	-	0.81	-	T = 23-30°C
33,217	CSTR-SG	13,000	26,000	0.65	1.30	20	600	360	840	500	69.2	24.3	6.1	0.5	2.9	A; T = 34°C
		13,000	26,000	1.30	2.60	10	600	390	910	570	72.8	23.4	3.0	0.5	2.9	A; T = 34°C
		13,000	26,000	2.60	5.20	5	690	420	880	540	75.4	23.4	1.0	0.5	2.9	A; T = 34°C
98,220	PFR-AG	-	12,900	-	1.29	10	-	1020	-	1450	70.0	22.0	8.0	~0.45	2.8	T = 35°C
	CSTR-SG	-	12,900	-	0.65	20	-	335	-	450	74.3	22.1	3.6	~0.45	2.8	A; T = 35°C
		-	12,900	-	0.65	20	-	390	-	525	74.3	22.1	3.6	~0.45	2.8	A; T = 35°C
		-	12,900	-	1.29	10	-	310	-	430	74.3	22.1	3.6	~0.45	2.8	A; T = 35°C
		-	12,900	-	0.65	20	-	295	-	415	71.3	14.7	14.0	~0.45	2.8	A; T = 20°C
		-	12,900	-	0.65	20	-	320		420	75.7	20.8	3.5	~0.45	2.8	Lime treated; T = 35°C
44-47	CSTR-AG	-	19,500	-	0.83	42	-	975	-	1250	78	-	-	-	3.5	T = 23°C;1:20 recycle
		-	39,000	-	0.98	74	-	475	-	610	78	-	-	-	3.5	T = 23°C;1:35 recycle
143	PFR-AG	2700	3600	1.8	2.4	1.5	-	-	265	190	-	-	-	0.75	-	T = 25°C; Batch
		2700	3600	0.6	0.8	4.5	-	-	0	0	-	-	-	0.75	-	T = 11°C; Batch
		2700	3600	1.2	1.6	2.25	-	-	0	0	-	-	-	0.75	-	T = 11°C; Batch
		24,500	38,800	0.34	0.53	73	-	-	690	440	-	-	-	0.63	-	T = 23°C; Continuous
		24,500	38,800	0.94	1.5	26	-	-	0	0	-	-	-	0.63	-	T = 23°C; Continuous
		24,500	38,800	0.94	1.5	26	-	-	0	0	-	-	-	0.63	-	T = 23°C; Continuous
		5950	9100	0.21	0.33	28	-	-	500	350	-	-	-	0.65	-	T = 23°C; Continuous
		5950	9100	0.40	0.61	15	-	-	440	300	-	-	-	0.65	-	T = 23°C; Continuous
		5950	9100	0.50	0.76	12	-	-	370	260	-	-	-	0.65	-	T = 23°C; Continuous
		5950	9100	0.72	1.1	8.3	-	-	390	260	-	-	-	0.65	-	T = 23°C; Continuous
		5950	9100	0.50	0.76	12	-	-	-	295	-	-	-	0.65	-	T = 23°C; Continuous
		5950	9100	0.33	0.51	18	-	-	520	355	-	-	-	0.65	-	T = 23°C; Continuous
		5950	9100	1.04	1.6	5.7	-	-	250	170	-	-	-	0.65	-	T = 23°C; Continuous
151	CSTR-SG	1930	2470	0.10	0.17	15	805	520	1080	700	74.6	18.3	7.1	0.62	3.0	T = 37°C; A; Batch
		1840	2480	0.19	0.25	10	-	-	640	490	-	-	-	0.78	2.8	T = 37°C; A; Batch
		1900	2900	0.19	0.29	10	540	390	635	460	85	13	1.6	0.66	2.6	T = 37°C; A; Batch
		1300	2740	0.13	0.27	10	490	300	580	360	84	13	2.7	0.47	3.3	T = 37°C; A; Batch
		3400	5300	0.68	1.06	5	510	350	620	425	82	15.4	2.0	0.64	2.4	T = 37°C; A; Batch
		3600	6120	3.6	6.1	1	-	-	290	170	-	-	-	0.59	~2.6	T = 37°C; A; Batch
		4800	6690	14	20	0.33	37	21	44	25	83	13	4	0.72	2.9	T = 37°C; A; Continuous
		4200	6300	25	39	0.17	45	60	56	75	80	15	5	0.67	2.8	T = 37°C; A; Continuous
		3880	6200	37	60	0.10	-	-	0	0	-	-	-	0.63	2.7	T = 37°C; A; Continuous
14	PFR-SG	-	25,000	-	4.0	6.6		-	-	520	-	-	-	-	>3.0	T = 33°C
		-	32,000	-	4.9	6.4		-	-	420	-	-	-	-	>3.0	T = 20°C
135,136	CSTR-SG	3940	7350	0.20	0.37	20	-	-	750	350	~70	-	-	0.54	5.8	T = 35°C
		3940	7350	0.26	0.49	15	-	-	820	450	~70	-	-	0.54	5.8	T = 35°C
		3940	7350	0.39	0.74	10	-	-	580	380	~70	-	-	0.54	5.8	T = 35°C
223	PFR-AG	2600	3600	1.4	1.8	1.8	-	-	-	887	-	-	-	0.81	-	T = 24°C, lime treated
30	CSTR-SG	1000	-	0.03	-	30	-	440	-	330	75	25	-	-	-	T = 24°C
		4000	-	0.13	-	30	-	440	-	330	75	25	-	-	-	T = 24°C
		10000	-	0.33	-	30	-	440	-	330	75	25	-	-	-	T = 24°C
237	PFR-AG	950	1870	3.8	6.8	0.25	-	305	-	350	79	10	-	0.51	2.6	T = 25°C
		950	1870	1.9	2.7	0.5	-	355	-	445	77	9	-	0.51	2.6	T = 25°C
		950	1870	1.0	1.4	1	-	315	-	385	82	12	-	0.51	2.6	T = 25°C
		950	1870	0.5	0.7	2	-	340	-	425	83	9	-	0.51	2.6	T = 25°C
		950	1870	0.3	0.5	3	-	285	-	375	82	7	-	0.51	2.6	T = 25°C
		950	1870	0.5	0.7	2	-	305	-	370	84	6	-	0.51	2.6	T = 25°C
		950	1870	0.3	0.6	3	-	320	-	380	82	8	-	0.51	2.6	T = 25°C

- Data not given
A = Nutrient Adjusted, BOD$_5$:N:P = 100:5:1

TABLE A-12 **Bench-Scale** Experimental Data for the Anaerobic Process for Heavy Metal Removal

			Influent Concentration, mg/l							Removal, %							
Reference	Process	pH	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Comments
14	PFR-SG	7.9	0.03	0.32	0.3	-	0.12	0.43	26	0	40	77	-	0	60	98	T = 20°C
			-	-	-	-	-	1.2	16	-	-	-	-	-	10	95	T = 33°C
33,217	CSTR-SG	-	0.1	0.22	0.03	600	0.76	0.19	65	>99	45	40	80	50	86	95	T = 34°C
68,70,73	CSTR-AG	7.4	0.03	1.7	5.6	430	0.38	1.2	16	52	91	88	97	84	84	94	T = 23°C
143	PFR-AG	7	0.01	0.45	0.3	245	-	0.70	5.0	0	0	50	94	-	71	80	T = 25°C
		7.5-8.0	0.03	1.0	1.30	810	1.4	1.2	155	0	90	50	99	0	67	>99	T = 23°C
		7.5-8.0	0.03	1.0	1.30	810	1.4	1.2	155	0	90	38	98	0	67	98	T = 23°C
		7.5-8.0	0.03	~1.0	1.30	810	1.4	1.2	155	0	90	69	93	0	83	98	T = 23°C
		7.5-8.0	0.03	1.0	1.30	810	1.4	1.2	155	0	90	69	97	0	67	>99	T = 23°C
		7.5-8.0	0.03	1.0	1.30	810	1.4	1.2	155	0	90	-	96	0	75	99	T = 23°C
		7.5-8.0	0.03	1.0	1.30	810	1.4	1.2	155	0	90	72	98	0	75	>99	T = 23°C
		7.5-8.0	0.03	1.0	1.30	810	1.4	1.2	155	0	90	46	96	0	75	99	T = 23°C
151	CSTR-SG	7.1-7.5	-	-	-	336	-	-	6	-	-	-	97	-	-	>90	T = 37°C
147	PFR-AG	7.3-8.0	0.01	-	0.05	36	0.05	-	0.19	0	-	0	62	92	-	0	T = 25°C

- Data not given

TABLE A-13. Bench-Scale Experimental Data for the Anaerobic Process for Alkali and Alkaline Earth Metal Removal

			Influent Concentration, mg/l					Removal, %					
Reference	Process	pH	Ca	Mg	Mn	K	Na	Ca	Mg	Mn	K	Na	Comments
33,217	CSTR-SG	-	1330	120	18	530	530	31	10	69	6	4	T = 34°C. Added lime to raise pH.
151	CSTR-SG	7.1-7.5	315	70	6.2	347	313	30	7	92	0	0	T = 37°C

- Data not given

APPENDIX B

PHYSICAL/CHEMICAL TREATMENT DATA

TABLE B-1. Bench-Scale Experimental Data for Chemical Coagulant Addition

Reference	Coagulant*	Dose, mg/l	pH		COD, mg/l		TOC, mg/l		Removal,%		Sludge, ml/l	Leachate Type**
			Initial	Final	Influent	Effluent	Influent	Effluent	COD	TOC		
19,20	Alum	10-1000	6.0	7.1	9100	9100	-	-	0	-	50 @ 500 mg/l	R
	$FeCl_3$	100-1000	6.0	5-7	9100	>7740	-	-	415	-	20 @ 500 mg/l	R
53,54,97	Alum+Lime	Alum=600 Lime=1640	5.4	8.0	17,000	14,800	-	-	13.3	-	-	R
	$FeCl_3$+$FeSO_4$+Lime	$FeCl_3$=1000; Lime=1640 $FeSO_4$ = 1450	5.4	8.0	17,000	15,100	-	-	11.5	-	-	R
	Polymer+Lime	Polymer=15 Lime=1000	5.4	8.0	17,000	15,100	-	-	11.5	-	-	R
133	Alum -	Not given	7.6	6.8	100(BOD_5)	25	-	-	-	-	4	R
134	Alum	10	7.0	7.1	9100	8700	-	-	4.4	-	3	R
	Alum	50	7.0	7.1	9100	8400	-	-	7.7	-	5	R
	Alum	100	7.0	7.1	9100	9100	-	-	0	-	85	R
	Alum	500	6.4	7.1	9100	8700	-	-	4.4	-	50	R
	Alum	1000	6.0	7.1	9100	8600	-	-	5.5	-	130	R
	$FeCl_3$	100	6.8	6	9100	8100	-	-	11	-	10	R
	$FeCl_3$	500	6.3	6	9100	8400	-	-	7.7	-	21	R
	$FeCl_3$	1000	5.9	6	9100	8700	-	-	4.4	-	33	R
	$FeCl_3$	1000	5.9	5	9100	8400	-	-	7.7	-	67	R
	$FeCl_3$	1000	5.9	7	9100	7800	-	-	14	-	28	R
	$FeSO_4$+Lime	760+600	6.4	7.0	-	-	1750	1500	-	15	120	R
	$FeSO_4$+Lime	760+1700	6.4	8.5	-	-	1750	1490	-	15	160	R
	$FeSO_4$+Lime	1360+0	6.4	6.3	-	-	1750	1520	-	13	2	R
	$FeSO_4$+Lime	1760+660	6.4	7.0	-	-	1750	1570	-	10	13	R
28	Alum	1000	7.3	7.3	10,650	9780	3120	2800	8	10	90	R
	Alum	2500	7.3	7.0	10,650	10,230	3120	2750	4	12	150	R
	Alum	5000	7.3	7.1	10,650	10,160	3120	2500	5	20	130	R
	Alum	6000	7.3	7.0	10,650	9770	3120	2320	8	26	130	R
	Alum	7000	7.3	7.1	10,650	9990	3120	2540	6	19	130	R
	Alum	8000	7.3	7.1	10,650	10,200	3120	2570	4	18	125	R
	Alum	9000	7.3	7.0	10,650	10,100	3120	2550	5	18	100	R
	Alum	10,000	7.3	7.1	10,650	10,200	3120	2550	5	18	120	R
	$FeCl_3$	1000	7.2	7.1	10,700	9900	3100	2550	7	18	190	R
	$FeCl_3$	1500	7.2	7.0	10,700	9940	3100	2570	7	19	220	R
	$FeCl_3$	2000	7.2	7.1	10,700	9720	3100	2520	9	19	250	R
	$FeCl_3$	2500	7.2	6.9	10,700	9560	3100	2480	11	20	275	R
236	$FeCl_3$	100	6.6	7.3	1240	1160	430	410	6	5	-	R
	$FeCl_3$	150	6.6	7.3	1240	1400	430	400	0	7	50	R
	$FeCl_3$	200	6.6	7.3	1240	1410	430	410	0	5	-	R
	Alum	90	6.6	6.3	1240	1100	430	380	11	12	-	R
	Alum	135	6.6	6.2	1240	1090	430	390	12	9	30	R
	Alum	180	6.6	6.1	1240	1300	430	390	0	9	-	R
	Alum	90	6.6	7.4	1240	1250	430	410	0	5	-	R
	Alum	135	6.6	7.4	1240	1080	430	410	13	5	-	R
	Alum	180	6.6	7.2	1240	1100	430	410	11	5	-	R
158	$FeCl_3$	Did not Specify 450-3150 mg/l pH_{opt}=5.4	-	-	11,600	-	-	-	88	-	-	R
	$FeCl_3$		-	-	4380	-	-	-	97	-	-	R
	$FeCl_3$		-	-	1570	-	-	-	83	-	-	R
	$FeCl_3$		-	-	320	-	-	-	64	-	-	R
	$FeCl_3$		-	-	690	-	-	-	39	-	-	R
	$FeCl_3$		-	-	6480	-	-	-	88	-	-	R
	$FeCl_3$		-	-	1200	-	-	-	79	-	-	R
	$FeCl_3$		-	-	350	-	-	-	66	-	-	R
	$FeCl_3$		-	-	520	-	-	-	52	-	-	R
143	Alum	75	-	7.7	530	480	-	-	10	-	-	R
	Alum	130	-	7.7	530	460	-	-	13	-	-	R
	Alum	200	-	7.7	530	490	-	-	8	-	-	R
	Alum	75	-	6	530	490	-	-	8	-	-	R
	Alum	130	-	6	530	475	-	-	10.5	-	-	R
	Alum	200	-	6	530	405	-	-	8.5	-	-	R
	$FeCl_3$	90	-	7.5	530	450	-	-	15	-	-	R
	$FeCl_3$	150	-	7.5	530	480	-	-	10	-	-	R
	$FeCl_3$	240	-	7.5	530	465	-	-	12.5	-	-	R
	$FeCl_3$	90	-	9	530	470	-	-	12	-	-	R
	$FeCl_3$	150	-	9	530	480	-	-	9.5	-	-	R
	$FeCl_3$	240	-	9	530	465	-	-	12.5	-	-	R
	Alum	75	-	7.6	400	355	-	-	11.3	-	-	B
	Alum	130	-	7.4	400	360	-	-	10	-	-	B
	Alum	200	-	6.6	400	-	-	-	-	-	-	B
	Alum	75	-	6	400	355	-	-	11.3	-	-	B
	Alum	130	-	6	400	355	-	-	11.3	-	-	B
	$FeCl_3$	90	-	7	400	355	-	-	11.3	-	-	B
	$FeCl_3$	150	-	7	400	355	-	-	11.3	-	-	B
	$FeCl_3$	90	-	9	400	350	-	-	12.5	-	-	B
	$FeCl_3$	150	-	9	400	350	-	-	12.5	-	-	B
	Alum	75	-	6	170	160	50	43	5.3	14	-	B
	Alum	130	-	6	170	125	50	38	26	24	-	B
	Alum	200	-	6	170	125	50	35	26	30	-	B
	$FeCl_3$	90	-	9	170	153	50	45	8.9	10	-	B
	$FeCl_3$	150	-	9	170	137	50	43	19	14	-	B
	$FeCl_3$	240	-	9	170	135	50	45	21	10	-	B
151	Alum+Lime	1000+530	6.4	7.1	-	-	1750	1600	-	8.9	85	R
	Alum+Lime	1400+0	6.4	5.5	-	-	1750	1590	-	9.0	80	R
	Alum+Lime	1400+660	6.4	7.0	-	-	1750	1510	-	14	100	R
	Alum+Lime	1400+1850	6.4	8.5	-	-	1750	1500	-	14	100	R
	Alum+Lime	1750+930	6.4	7.1	-	-	1750	1490	-	15	130	R
	Alum+Lime	2250+1060	6.4	7.1	-	-	1750	1050	-	40	190	R
	$FeSO_4$+Lime	0+165	6.4	7.0	-	-	1750	1710	-	2	20	R
	$FeSO_4$+Lime	270+400	6.4	7.0	-	-	1750	1550	-	12	40	R
	$FeSO_4$+Lime	550+530	6.4	7.2	-	-	1750	1480	-	15	100	R
96	Alum	30	5.4	8.2	35,000	33,500	-	-	4	-	-	R
	Alum	45	5.4	8.2	35,000	30,000	-	-	14	-	-	R
	Alum	55	5.4	8.2	35,000	28,000	-	-	20	-	-	R
	Alum	65	5.4	8.2	35,000	27,000	-	-	23	-	-	R
	Alum	75	5.4	8.2	35,000	27,500	-	-	22	-	-	R
	Alum	90	5.4	8.2	35,000	28,500	-	-	20	-	-	R
	Alum	30	5.5	8.2	34,000	32,500	-	-	4	-	-	R
	Alum	45	5.5	8.2	34,000	30,500	-	-	10	-	-	R
	Alum	55	5.5	8.2	34,000	29,500	-	-	13	-	-	R
	Alum	65	5.5	8.2	34,000	28,500	-	-	16	-	-	R
	Alum	75	5.5	8.2	34,000	29,000	-	-	15	-	-	R
	Alum	90	5.5	8.2	34,000	29,000	-	-	15	-	-	R
	Alum	30	5.5	8.2	33,000	32,000	-	-	3	-	-	R
	Alum	45	5.5	8.2	33,000	30,000	-	-	9	-	-	R
	Alum	55	5.5	8.2	33,000	29,000	-	-	12	-	-	R
	Alum	65	5.5	8.2	33,000	28,000	-	-	15	-	-	R
	Alum	75	5.5	8.2	33,000	29,000	-	-	12	-	-	R
	Alum	90	5.5	8.2	33,000	29,000	-	-	12	-	-	R

- Data not given
* Alum as $Al_2(SO_4)_3$
Lime as $Ca(OH)_2$
** R = Raw leachate
B = Biologically treated effluent

TABLE B-2. Bench-Scale Experimental Data for Chemical Precipitant Addition

Reference	Precipitant	Dose, mg/l	pH		COD, mg/l		TOC, mg/l		Removal,%		Sludge, ml/l	Leachate Type**
			Initial	Final	Influent	Effluent	Influent	Effluent	COD	TOC		
17,18,176	Lime*	2350	5.3	6.9	14,000	9200	5200	2700	34	48	-	R
19,20	Lime	750-1750	6.0	8.5-12	10,650	10,650	-	-	0	-	250 @ 1000mg/l	R
	Na_2S	10-1000	6.0	6.0-6.3	10,650	10,650	-	-	0	-	50 @ 750mg/l	R
44,45,70	Lime	1000-8000	7.4	12	-	-	700	520	-	26@8000 mg/l	150	B
	Lime	1500-4000	9	12	700	560	-	-	20@4000mg/l	-	140	B
53,54,97	Lime	2760	5.4	11.0	17,000	14,900	-	-	13	-	-	R
	NaOH	2660	5.4	11.0	17,000	15,400	-	-	9.8	-	-	R
134	Lime	870	6.0	9.0	10,700	10,600	-	-	1.0	-	130	R
	Lime	1000	6.0	9.5	10,700	10,400	-	-	2.8	-	200	R
	Lime	1150	6.0	10.0	10,700	9970	-	-	6.8	-	245	R
	Lime	1280	6.0	10.5	10,700	10,300	-	-	3.7	-	250	R
	Lime	1390	6.0	11.0	10,700	10,700	-	-	0	-	250	R
	Lime	1600	6.0	11.5	10,700	10,100	-	-	5.6	-	275	R
	Lime	1840	6.0	12.0	10,700	10,400	-	-	2.8	-	280	R
	Lime	1060	7.8	9.0	560	560	-	-	0	-	6	B
	Lime	2700	7.8	11.0	560	515	-	-	8.0	-	26	B
	Lime	470	9.0	10.0	370	370	-	-	0	-	5	B
	Lime	1400	9.0	11.5	370	260	-	-	30	-	68	B
	Na_2S	10	6.0	6.0	10,700	10,200	-	-	4.7	-	30	R
	Na_2S	25	6.0	6.0	10,700	10,000	-	-	6.5	-	30	R
	Na_2S	50	6.0	6.0	10,700	10,700	-	-	0	-	30	R
	Na_2S	100	6.0	6.1	10,700	10,200	-	-	4.7	-	35	R
	Na_2S	500	6.0	6.3	10,700	10,170	-	-	4.9	-	50	R
	Na_2S	1000	6.0	6.4	10,700	10,600	-	-	1.0	-	65	R
143	Lime	pH 9	-	9	530	490	-	-	7	-	-	R
	Lime	pH 10	-	10	530	480	-	-	10	-	-	R
	Lime	pH 11	-	11	530	445	-	-	16	-	-	R
	Lime	pH 12	-	12	530	440	-	-	17	-	-	R
	Lime	pH 9	-	9	400	370	-	-	7.5	-	-	B
	Lime	pH 10	-	10	400	370	-	-	7.5	-	-	B
	Lime	pH 11	-	11	400	360	-	-	10	-	-	B
	Lime	pH 12	-	12	400	355	-	-	11.3	-	-	B
	Lime	pH 9	-	9	170	113	50	53	3.0	0	-	B
	Lime	pH 10	-	10	170	147	50	48	13	4	-	B
	Lime	pH 11	-	11	170	135	50	45	20	10	-	B
	Lime	pH 12	-	12	170	104	50	35	38	30	-	B
265	Lime	150	6.3	6.6	5030	4620	-	-	8.2	-	42	R
	Lime	300	6.3	7.2	5030	4350	-	-	14	-	104	R
	Lime	450	6.3	7.9	5030	4280	-	-	15	-	154	R
	Lime	600	6.3	8.3	5030	4380	-	-	13	-	188	R
	Lime	750	6.3	9.4	5030	4340	-	-	13	-	220	R
	Lime	900	6.3	9.7	5030	4240	-	-	16	-	276	R
	Lime	1050	6.3	10.3	5030	4140	-	-	18	-	232	R
	Lime	1200	6.3	10.9	5030	3930	-	-	22	-	412	R
	Lime	1350	6.3	11.2	5030	3850	-	-	24	-	420	R
	Lime	1500	6.3	11.5	5030	-	-	-	-	-	440	R
	Lime	150	5.3	6.2	12,900	12,200	-	-	5.4	-	60	R
	Lime	300	5.3	6.5	12,900	11,600	-	-	10	-	60	R
	Lime	450	5.3	6.8	12,900	11,400	-	-	12	-	80	R
	Lime	600	5.3	7.0	12,900	10,800	-	-	16	-	180	R
	Lime	700	5.3	6.9	12,900	10,700	-	-	17	-	200	R
	Lime	750	5.3	7.1	12,900	10,200	-	-	21	-	240	R
	Lime	800	5.3	7.0	12,900	10,500	-	-	19	-	280	R
	Lime	900	5.3	7.2	12,900	10,050	-	-	22	-	300	R
	Lime	1000	5.3	7.2	12,900	9730	-	-	24	-	300	R
	Lime	1100	5.3	7.3	12,900	9580	-	-	26	-	340	R
	Lime	1200	5.3	7.4	12,900	9600	-	-	26	-	300	R
118,119,261	Lime	100	3.0	8.2	400	385	-	-	4	-	-	B
	Lime	500	8.0	10.5	400	360	-	-	10	-	-	B
	Lime	900	8.0	11.5	400	300	-	-	25	-	-	B
	Lime	1500	8.0	11.8	400	260	-	-	35	-	-	B
28	Lime	1000	7.1	7.6	10,660	10,450	3290	3170	2	4	50	R
	Lime	2500	7.1	8.1	10,660	10,600	3290	3170	<1	4	120	R
	Lime	5000	7.1	8.4	10,660	10,080	3290	3200	6	3	150	R
	Lime	6000	7.1	9.7	10,660	9800	3290	2010	8	9	115	R
	Lime	7000	7.1	10.1	10,660	9580	3290	3150	10	4	180	R
	Lime	8000	7.1	10.4	10,660	9720	3290	3080	9	6	260	R
	Lime	9000	7.1	11.7	10,660	9570	3290	2730	10	17	475	R
	Lime	10,000	7.1	12.1	10,660	9620	3290	3010	10	9	470	R
238	Lime	190	6.6	7.0	1240	1210	430	430	2	0	-	R
	Lime	190	6.6	7.0	1240	1190	430	420	4	2	-	R
	Lime	225	6.6	7.9	1240	1030	430	370	9	14	-	R
	Lime	225	6.6	7.9	1240	1010	430	370	10	14	-	R
	NaOH	200	6.6	7.7	1240	1160	430	410	6	5	-	R
	NaOH	240	6.6	7.8	1240	1160	430	420	6	2	-	R
239	Lime	6,000	5.8	12.2	22,900	20,700	9850	9250	10	6	250	R

- = Data not given
* Lime as $Ca(OH)_2$

** R = Raw leachate
B = Biologically treated effluent

TABLE B-3. Bench-Scale Experimental Data for Heavy Metal Removal for the Chemical Addition Processes.

				Influent Concentration, mg/l							Removal, %							Leachate
Reference	Chemical	Dose, mg/l	pH	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Cd	Cr	Cu	Fe	Pb	Ni	Zn	Type**
17,18,176	$Ca(OH)_2$	2350	6.9	-	-	0.39	47	-	-	12.5	-	-	96	99	-	-	>99	R
	O_3	180-250	6.9	-	-	0.39	47	-	-	12.5	-	-	96	99	-	-	>99	R
143	$Al_2(SO_4)_3$	75	7.7	-	-	-	85	-	-	0.82	-	-	-	99	-	-	>90	R
	$Al_2(SO_4)_3$	75	6	-	-	-	85	-	-	0.82	-	-	-	99		-	>90	R
	$FeCl_3$	90	7.5	-	-	-	85	-	-	0.82	-	-	-	99	-	-	>90	R
	$FeCl_3$	90	9	-	-	-	85	-	-	0.82	-	-	-	99	-	-	>90	R
	$Ca(OH)_2$	pH 9	9	-	-	-	85	-	-	0.82	-	-	-	99	-	-	>90	R
	$Ca(OH)_2$	pH 11	11	-	-	-	85	-	-	0.82	-	-	-	99	-	-	>90	R
	$Al_2(SO_4)_3$	75	7.5	-	-	-	21	-	-	0.37	-	-	-	97	-	-	>75	B
	$Al_2(SO_4)_3$	75	6.0	-	-	-	21	-	-	0.37	-	-	-	97	-	-	>75	B
	$FeCl_3$	90	7.0	-	-	-	21	-	-	0.37	-	-	-	93	-	-	32	B
	$FeCl_3$	90	9.0	-	-	-	21	-	-	0.37	-	-	-	98	-	-	>75	B
	$Ca(OH)_2$	pH 9	9	-	-	-	21	-	-	0.37	-	-	-	83	-	-	>75	B
	$Ca(OH)_2$	pH 11	11	-	-	-	21	-	-	0.37	-	-	-	99	-	-	>75	B
151	$Ca(OH)_2$	165	7.0	-	-	-	26	-	-	30	-	-	-	0	-	-	98	R
	$Al_2(SO_4)_3$	1400	-	-	-	-	26	-	-	30	-	-	-	54	-	-	99	R
	$FeSO_4$	1360	6.3	-	-	-	26	-	-	30	-	-	-	0	-	-	99	R
229	O_3 -	240τ=4 hrs	8.8	-	-	-	-	-	73	-	-	-	-	-	-	6	-	R
133	$Al_2(SO_4)_3$	-	6.8	-	-	0.56	20	0.10	-	-	-	-	89	93	20	-	-	R
26	$Ca(OH)_2$	1650	10.5-11.0	0.05	-	-	1000	-	-	8	20	-	-	99	-	-	>99	R;Full Scale
19,20	$Ca(OH)_2$	750	8.5	-	-	-	330	-	-	-	-	-	-	>99	-	-	-	D(1:1)
	Na_2S	1000	8.5	-	-	-	330	-	-	-	-	-	-	>99	-	-	-	D(1:1)
	Cl_2	400	7.0	-	-	-	3.7	-	-	-	-	-	-	95	-	-	-	D(1:25)
	$KMnO_4$	500	5.8	-	-	-	85	-	-	-	-	-	-	99	-	-	-	R
	$Al_2(SO_4)_3$	1000	7.1	-	-	-	330	-	-	-	-	-	-	97	-	-	-	D(1:1)
	$FeCl_3$	1000	7.0	-	-	-	330	-	-	-	-	-	-	98	-	-	-	D(1:1)
32	$Ca(OH)_2$	2000	8.0	-	-	-	21	-	-	-	-	-	-	52	-	-	-	R
	$Ca(OH)_2+FeCl_3$	2500+200	8.8	-	-	-	21	-	-	-	-	-	-	93	-	-	-	R
134	$Ca(OH)_2$	870	9.0	-	-	-	330	-	-	-	-	-	-	99	-	-	-	D(1:1)
	$Ca(OH)_2$	1020	9.5	-	-	-	330	-	-	-	-	-	-	>99	-	-	-	D(1:1)
	Na_2S	500	6.3	-	-	-	330	-	-	-	-	-	-	18	-	-	-	D(1:1)
	Na_2S	1000	6.4	-	-	-	330	-	-	-	-	-	-	79	-	-	-	D(1:1)
	$Al_2(SO_4)_3$	100	6.9	-	-	-	85	-	-	-	-	-	-	60	-	-	-	R
	$Al_2(SO_4)_3$	500	6.4	-	-	-	85	-	-	-	-	-	-	94	-	-	-	R
	$FeCl_3$	500	6	-	-	-	85	-	-	-	-	-	-	55	-	-	-	R
	$FeCl_3$	1000	6	-	-	-	85	-	-	-	-	-	-	62	-	-	-	R
	$FeCl_3$	1000	7	-	-	-	85	-	-	-	-	-	-	98	-	-	-	R
	Cl_2	800	7.0	-	-	-	3.7	-	-	-	-	-	-	99	-	-	-	D(1:25)
	$Ca(OCl)_2$	1000	7.0	-	-	-	38	-	-	-	-	-	-	99	-	-	-	D(1:1)
	$KMnO_4$	50	5.8	-	-	-	330	-	-	-	-	-	-	45	-	-	-	D(1:1)
	$KMnO_4$	100	5.8	-	-	-	330	-	-	-	-	-	-	77	-	-	-	D(1:1)
	$KMnO_4$	500	5.8	-	-	-	330	-	-	-	-	-	-	99	-	-	-	D(1:1)
	O_3	5000 @ 1 hr	7.8	-	-	-	45	-	-	-	-	-	-	82	-	-	-	R
	O_3	5000 @ 4 hr	7.5	-	-	-	45	-	-	-	-	-	-	96	-	-	-	R
265	$Ca(OH)_2$	150	6.6	-	-	-	115	-	-	-	-	-	-	21	-	-	-	R
	$Ca(OH)_2$	300	7.2	-	-	-	115	-	-	-	-	-	-	70	-	-	-	R
	$Ca(OH)_2$	450	7.9	-	-	-	115	-	-	-	-	-	-	99	-	-	-	R
	$Ca(OH)_2$	300	6.5	-	-	-	790	-	-	-	-	-	-	26	-	-	-	R
	$Ca(OH)_2$	900	7.2	-	-	-	790	-	-	-	-	-	-	70	-	-	-	R
28	$Ca(OH)_2$	1000	7.6	-	-	-	220	-	-	-	-	-	-	87	-	-	-	R
	$Ca(OH)_2$	2500	8.1	-	-	-	220	-	-	-	-	-	-	91	-	-	-	R
	$Ca(OH)_2$	5000	8.4	-	-	-	220	-	-	-	-	-	-	91	-	-	-	R
	$Ca(OH)_2$	6000	9.7	-	-	-	220	-	-	-	-	-	-	96	-	-	-	R
	$Ca(OH)_2$	7000	10.1	-	-	-	220	-	-	-	-	-	-	>99	-	-	-	R
	$Ca(OH)_2$	8000	10.4	-	-	-	220	-	-	-	-	-	-	>99	-	-	-	R
	$Ca(OH)_2$	9000	11.7	-	-	-	220	-	-	-	-	-	-	>99	-	-	-	R
	$Ca(OH)_2$	10,000	12.1	-	-	-	220	-	-	-	-	-	-	>99	-	-	-	R
	$FeCl_3$	1000	7.1	-	-	-	150	-	-	-	-	-	-	78	-	-	-	R
	$FeCl_3$	1500	7.0	-	-	-	150	-	-	-	-	-	-	77	-	-	-	R
	$FeCl_3$	2000	7.1	-	-	-	150	-	-	-	-	-	-	82	-	-	-	R
	$FeCl_3$	2500	6.9	-	-	-	150	-	-	-	-	-	-	79	-	-	-	R
	$Al_2(SO_4)_3$	1000	7.3	-	-	-	165	-	-	-	-	-	-	86	-	-	-	R
	$Al_2(SO_4)_3$	2500	7.0	-	-	-	165	-	-	-	-	-	-	87	-	-	-	R
	$Al_2(SO_4)_3$	5000	7.1	-	-	-	165	-	-	-	-	-	-	96	-	-	-	R
28	$Al_2(SO_4)_3$	6000	7.0	-	-	-	165	-	-	-	-	-	-	96	-	-	-	R
	$Al_2(SO_4)_3$	7000	7.1	-	-	-	165	-	-	-	-	-	-	97	-	-	-	R
	$Al_2(SO_4)_3$	8000	7.1	-	-	-	165	-	-	-	-	-	-	96	-	-	-	R
	$Al_2(SO_4)_3$	9000	7.0	-	-	-	165	-	-	-	-	-	-	94	-	-	-	R
	$Al_2(SO_4)_3$	10,000	7.1	-	-	-	165	-	-	-	-	-	-	97	-	-	-	R
238	$FeCl_3$	100	7.3	-	0.064	0.07	84	-	-	3.0	-	53	21	98	-	-	97	R
	$FeCl_3$	150	7.3	-	0.064	0.07	84	-	-	3.0	-	53	36	98	-	-	97	R
	$FeCl_3$	200	7.3	-	0.064	0.07	84	-	-	3.0	-	53	21	77	-	-	97	R
	Alum	90	6.3	-	0.064	0.07	84	-	-	3.0	-	53	7	98	-	-	91	R
	Alum	135	6.2	-	0.064	0.07	84	-	-	3.0	-	53	0	>99	-	-	89	R
	Alum	180	6.1	-	0.064	0.07	84	-	-	3.0	-	53	0	>99	-	-	93	R
	Alum	90	7.4	-	0.064	0.07	84	-	-	3.0	-	53	43	>99	-	-	95	R
	Alum	135	7.4	-	0.064	0.07	84	-	-	3.0	-	53	21	>99	-	-	96	R
	Alum	180	7.2	-	0.064	0.07	84	-	-	3.0	-	53	21	>99	-	-	97	R
	$Ca(OH)_2$	190	7.0	-	0.064	0.07	84	-	-	3.0	-	53	14	1	-	-	43	R
	$Ca(OH)_2$	190	7.0	-	0.064	0.07	84	-	-	3.0	-	30	7	12	-	-	0	R
	$Ca(OH)_2$	225	7.9	-	0.064	0.07	84	-	-	3.0	-	53	36	97	-	-	95	R
	$Ca(OH)_2$	225	7.9	-	0.064	0.07	84	-	-	3.0	-	53	64	98	-	-	97	R
	NaOH	200	7.7	-	0.064	0.07	84	-	-	3.0	-	53	64	91	-	-	88	R
	NaOH	240	7.8	-	0.064	0.07	84	-	-	3.0	-	53	64	99	-	-	94	R
	NaOH	200	7.8	-	0.08	0.035	80	-	-	3.5	-	-	-	95	-	-	-	R
	NaOH	300	8.0	-	0.08	0.035	80	-	-	3.5	-	63	57	94	-	-	91	R
	NaOH	360	8.0	-	0.08	0.035	80	-	-	3.5	-	63	43	97	-	-	95	R
96	Alum	65	8.2	-	-	-	500	-	-	-	-	-	-	98	-	-	-	R

- Data not given

* R = Raw leachate
B = Biologically treated leachate
D = Diluted raw leachate

TABLE B-4. **Bench-Scale Experimental Data for Alkali and Alkaline Earth Metal Removal for the Chemical Addition Processes**

Reference	Chemical	Dose, mg/l	pH	Influent Concentration, mg/l					Removal, %					Leachate Type*
				Ca	Mg	Mn	K	Na	Ca	Mg	Mn	K	Na	
17,18,176	$Ca(OH)_2$	2350	6.9	-	-	10	156	188	-	-	>99	27	43	R
	O_3	180-250	6.9	-	-	10	156	188	-	-	>99	27	43	R
32	$Ca(OH)_2$	2000	8.0	-	-	0.72	-	-	-	-	96	-	-	R
	$Ca(OH)_2 + FeCl_3$	2500+200	8.8	-	-	0.72	-	-	-	-	96	-	-	R
151	$Ca(OH)_2$	165	7.0	178	100	25	380	-	0	0	28	8	-	R
	$Al_2(SO_4)_3$	1400	-	178	100	25	380	-	0	60	28	16	-	R
	$FeSO_4$	1360	6.3	178	100	25	380	-	6	0	28	18	-	R
265	$Ca(OH)_2$	300	6.5	-	160	-	-	-	-	0	-	-	-	R
	$Ca(OH)_2$	900	7.2	-	160	-	-	-	-	16	-	-	-	R

- Data not given

*R = Raw leachate

TABLE B-5. Bench-Scale Experimental Data for Chemical Oxidant Addition

Reference	Oxidant	Dose, mg/l.	pH		COD, mg/l		TOC, mg/l		Removal, %		Sludge, ml/l	Leachate Type**
			Initial	Final	Influent	Effluent	Influent	Effluent	COD	TOC		
17,18,176	O_3	100 @ τ=30 min	5.3	6.9	14,000	9200	5200	2700	34	48	-	R
19,20	Cl_2	400-1540	7.0	7.0	340	260	-	-	25 @ 1200	-	6	D (1:25)
	$KMnO_4$	10-10,000	7.0	5.8	10,650	8500	-	-	20 @ 10,000	-	110 @ 500 mg/l	R
44,45,73	O_3	1.2-1.5% O_3 @ 4 l/min, τ= 3 hrs	8.8	-	670	300	250	120		48		B
53,54,97	NaOCl	3400 as NaOCl	8.4	9.8	330	220	-	-	33	-	-	B
	NaOCl	3000 as NaOCl	8.4	9.5	320	260	-	-	19	-	-	B
	NaOCl	2500 as NaOCl	7.6	8.9	270	120	-	-	56	-	-	B
	NaOCl	1600 as NaOCl	7.6	8.9	290	90	-	-	69	-	-	B
134	Cl_2	400	2.2	7.0	340	300	-	-	13	-	4.5	D (1:25)
	Cl_2	800	2.0	7.0	340	290	-	-	15	-	7.0	D (1:25)
	Cl_2	1200	1.8	7.0	340	260	-	-	24	-	5.0	D (1:25)
	Cl_2	1540	1.6	7.0	340	320	-	-	5.9	-	7.3	D (1:25)
	$Ca(OCl)_2$	1000	8.0	7.0	1500	1400	-	-	6.7	-	3	D (1:1)
	$Ca(OCl)_2$	2000	8.0	7.0	1500	1400	-	-	6.7	-	3	D (1:1)
	$Ca(OCl)_2$	4000	8.2	7.0	1500	1100	-	-	27	-	3	D (1:1)
	$Ca(OCl)_2$	8000	9.0	7.0	1500	760	-	-	49	-	4	D (1:1)
	$Ca(OCl)_2$	12,000	9.9	7.0	1500	900	-	-	40	-	4	D (1:1)
	$Ca(OCl)_2$	15,000	10.2	7.0	1500	1000	-	-	33	-	5	D (1:1)
	O_3	τ=1 hr @ $Q=\frac{26 mg O_3}{min}$	7.4	7.8	7160	6800	-	-	5.0	-	0	R
	O_3	τ=4 hr @ $Q=\frac{26 mg O_3}{min}$	7.4	7.5	7160	4500	-	-	37	-	0	R
134	$KMnO_4$	10	6.0	5.8	10,900	10,800	-	-	1	-	40	R
	$KMnO_4$	25	6.0	5.8	10,900	10,700	-	-	1.8	-	45	R
	$KMnO_4$	50	6.0	5.8	10,900	10,350	-	-	5.1	-	50	R
	$KMnO_4$	100	6.0	5.8	10,900	10,300	-	-	5.5	-	60	R
	$KMnO_4$	500	6.0	5.8	10,900	9800	-	-	10	-	120	R
	$KMnO_4$	1000	6.0	5.8	10,900	9700	-	-	11	-	-	R
	$KMnO_4$	2500	6.0	5.8	10,900	9600	-	-	12	-	-	R
	$KMnO_4$	5000	6.0	5.8	10,900	9350	-	-	14	-	-	R
	$KMnO_4$	7500	6.0	5.8	10,900	9100	-	-	17	-	-	R
	$KMnO_4$	10,000	6.0	5.8	10,900	8900	-	-	18	-	-	R
151	NaOCl	200 as Cl_2	-	-	-	-	1750	1590	-	9.1	-	R
	NaOCl	500 as Cl_2	-	-	-	-	1750	1510	-	14	-	R
	NaOCl	1000 as Cl_2	-	-	-	-	1750	1420	-	19	-	R
	NaOCl	2000 as Cl_2	-	-	-	-	1750	1360	-	22	-	R
229	O_3	24 @ τ=4hrs	8.8	-	7600	6300	-	-	17	-	-	R

- **Data not given**

* **R = Raw leachate**

B = Biologically treated effluent

D = Diluted raw leachate

TABLE B-6. Bench-Scale Experimental Data for COD and TOC Removal for the Physical Treatment Processes

Reference	Process*	Test	Leachate Type	pH	Influent Concentration, mg/l		Removal, %		Comments
					COD	TOC	COD	TOC	
44,45,70	RO		Raw	5.5	-	13,000	-	70	KP-98 Membrane; P=600 psig; Flux=5.5 gpd/ft^2
	RO		Raw	5.5	-	13,000	-	75	KP-98 Membrane; P=1500 psi; Flux=8.9
	RO		Raw	5.5	-	18,500	-	56	KP-98 Membrane; P=600 psi; Flux=3.7
	RO		Raw	5.5	-	18,500	-	59	KP-98 Membrane; P=1500 psi; Flux=6.2
	RO		Raw	5.5	-	13,000	-	85	NS-100 Membrane; P=600 psi; Flux=7
	RO		Raw	5.5	-	13,000	-	88	NS-100 Membrane; P=1500 psi; Flux=11
	RO		Raw	8.0	-	13,000	-	92	KP-98 Membrane; P=600 psi; Flux=6.1
	RO		Raw	8.0	-	13,000	-	93	KP-98 Membrane; P=1500 psi; Flux=10
	RO		Raw	8.0	-	18,500	-	89	KP-98 Membrane; P=600 psi; Flux=3.9
	RO		Raw	8.0	-	18,500	-	60	KP-98 Membrane; P=150 psi; Flux=7.1
	RO		Raw	8.0	-	13,000	-	93	NS-100 Membrane; P=600 psi; Flux=7.3
	RO		Raw	8.0	-	13,000	-	94	NS-100 Membrane; P=1500 psi; Flux=12.5
	RO		AL Effluent	8.8	-	214	-	95	NS-100 Membrane; P=600 psi; Flux=9.8
	RO		AC Effluent	8.8	-	48	-	86	NS-100 Membrane; P=600 psi; Flux=12.5
	RO		IX(-) Effluent	5.5	-	133	-	97	NS-100 Membrane; P=600 psi; Flux=12.0
	RO		IX(-) Effluent	5.5	-	119	-	94	NS-100 Membrane; P=600 psi; Flux=12.4
	RO		IX(-) Effluent	5.5	-	143	-	94	NS-100 Membrane; P=600 psi; Flux=11.9
	IX(-)**	Column	AL Effluent	8.8	500	200	6	31	Duolite A-7
		Column	AL Effluent	8.8	500	200	59	43	Amberlite IRA-938
		Column	AL Effluent	8.8	500	200	41	26	Amberlite XE-299 HP
		Column	AL Effluent	6.2	500	200	48	43	Duolite A7
	AC**	Column	AL Effluent	8.8	500	200	74	71	GAC (40x48)
207	IX(Mixed Resin)	Batch(2g/l)	AS Effluent	7.7	100	-	36	-	Dowex 50Wx8 H$^+$ and Dowex 1x8 OH$^-$; τ = 1 hour
		Batch(10g/l)	AS Effluent	5.0	100	-	68	-	Dowex 50Wx8 H$^+$ and Dowex 1x8 OH$^-$; τ = 1 hour
207	AC	Batch(4g/l)	IX Effluent	8.4	115	-	>99	-	τ = 30 min. Used 2g/l IX Effluent
	IX(-)	Batch(2g/l)	AS Effluent	7.3	185	-	10	-	Dowex 50Wx8 H$^+$; τ = 1 hour
		Batch(10g/l)	AS Effluent	2.9	185	-	19	-	Dowex 50Wx8 H$^+$; τ = 1 hour

*RO = Reverse Osmosis
IX = Ion Exchange; (-) = anionic exchange resin
AC = Activated Carbon Adsorption
**Removal after 50 bed volumes
psi = 6.895 kN/m^2
gpd/ft^2 = 0.041 m^3/m^2.d

TABLE B-7. Freundlich Isotherm Parameters for Activated Carbon Adsorption

			Freundlich Isotherm Parameters							
			COD			TOC				
Reference	AC Type & Size	Test	C_o,mg/l	$\frac{X}{M}$, $\frac{mgCOD}{gAC}$	1/n, $\frac{mgCOD}{gAC/mg/l}$	C_o,mg/l	$\frac{X}{M}$, $\frac{mgTOC}{gAC}$	1/n, $\frac{mgTOC}{gAC/mg/l}$	Leachate Type*	Comments
44,45,70	GAC	Batch	-	-	-	13,800	46	0.75-1.2	R	
	GAC	Batch	-	-	-	395	300	1.9-12	R	Older leachate
	GAC	Batch	-	-	-	120	68	0.81	RD	Diluted leachate
	GAC	Batch	630	520	0.93-1.7	225	174	-	AF(aerated)	
	GAC	Batch	830	261	0.70-2.3	320	102	-	AF	
	GAC	Column	-	-	-	120	38	-	RD	Breakthrough @ 200 BOD volumes
	GAC;40x48	Column	540	-	-	210	-	-	AL	τ=0.7 min; Max. COD&TOC Rem = 67%; After 50 BV= 56%
	GAC;40x48	Column	540	-	-	210	-	-	AL	τ=3.7 min: Max. COD&TOC Rem = 86%; After 50 BV = 74%
53,54,97	GAC;6x14	Column	330	-	-	140	-	-	AS	τ=15 min; COD Rem.=70%; TOC Rem.=78%
	GAC;6x14	Column	320	-	-	130	-	-	AS	τ=15 min; COD Rem.=47%; TOC Rem.=75%
	GAC;6x14	Column	270	-	-	76	-	-	AS	τ=15 min; COD Rem.=52%; TOC Rem.=53%
	GAC;6x14	Column	290	-	-	76	-	-	AS	τ=15 min; COD Rem.=55%; TOC Rem.=53%
134	PAC;325 mesh	Batch	5000	2500	9.5	-	-	-	R	
143	PAC	Batch	508	550	1.4	153	165	1.1	C	
	PAC	Batch	344	600	2.5	98	230	2.5	B+C	
	PAC	Batch	232	150	0.98	63	130	2.4	B+C	
	PAC	Batch	594	800	2.2	160	140	0.97	C	
	PAC	Batch	192	600	5.9	65	165	2.9	B+C	
151	PAC	Batch	-	-	-	2000	144	2.7	R	
207	PAC	Batch	184	540	1.57	-	-	-	AS	WV-Nuchar C-190-N
229	GAC;0.9 mm	Batch	6000	-	-	-	-	-	$C(O_3)$	COD_{eff}=3420 mg/l; 43% removal; 4 g/lAC
178,179			-	-	-	4200	200	0.60	R	Love Canal
28	GAC;12x40	Batch	2990	340	3.2	-	-	-	C	Filtrasorb 400, 1050-1200 m^2/g
	GAC;12x40	Batch	2950	300	3.0	-	-	-	C	WV-G Nuchar, 1100 m^2/g
	GAC;10x30	Batch	2930	290	2.6	-	-	-	C	Hydrodarco, 650 m^2/g
	GAC;12x40	Column	3000	-	-	-	-	-	C	22% Removal @ 50 BV; τ=4 min (1.55 gpm/ft^2)
	GAC;12x40	Column	3000	-	-	-	-	-	C	25% Removal @ 50 BV; τ=4 min (1.55 gpm/ft^2)
	GAC;10x30	Column	2960	-	-	-	-	-	C	14% Removal @ 50 BV; τ=4 min (1.55 gpm/ft^2)
	GAC;12x40	Column	1000	-	-	-	-	-	B	35% Removal @ 50 BV; τ=4 min (1.55 gpm/ft^2)
	GAC;12x40	Column	3000	-	-	-	-	-	C	35% Removal @ 50 BV; τ=10 min (0.65 gpm/ft^2)
	GAC;12x40	Batch	1010	0.14					B	

- Data not given
* R = Raw
RD = Raw Diluted
AS = AS Effluent
BV = Bed Volumes
AL = AL Effluent
AF = AF Effluent
C = Chemically Treated Effluent
B+C = Biologically + Chemically Treated Effluent
gpm/ft^2= 0.68 $l/m^2 \cdot s$

TABLE B-8. Bench-Scale Experimental Data for Heavy Metal Removal for the Physical Treatment Processes

Reference	Process*	Test	Leachate Type	pH	Influent Concentration, mg/l							Removal, %							Comments
					Cd	Cr	Cu	Fe	Pb	Ni	Zn	Cd	Cr	Cu	Fe	Pb	Ni	Zn	
151	AC	Batch(8g/l)	Raw	-	-	-	-	26	-	-	30	-	-	-	-	-	-	>99	PAC
229	AC	Batch(4g/l)	Ozonated Raw	8.8	-	-	-	-	-	60	-	-	-	-	96	-	37	-	Size: 0.88 mm
242,243	AC	Continuous (1 l/min)	Raw	7.1	0.026	0.07	0.24	22	0.23	0.13	0.69	27	0	0	10	22	0	0	GAC (6-14 mesh);
	IX	Continuous (1 l/min)	Raw	7.5	0.082	0.13	0.28	14	0.18	0.21	0.78	96	0	14	39	33	14	20	Glauconitic sand
134	AC	Batch(2g/l)	Raw	7.6	-	-	-	66	-	-	-	-	-	-	73	-	-	-	PAC (325 mesh)
	AC	Batch(8g/l)	Raw	7.6	-	-	-	66	-	-	-	-	-	-	96	-	-	-	PAC (325 mesh)
	AC	Batch(16g/l)	Raw	7.6	-	-	-	66	-	-	-	-	-	-	97	-	-	-	PAC (325 mesh)
	AC	Column (τ= 4 min)	Raw	7.6	-	-	-	40	-	-	-	-	-	-	65	-	-	-	GAC
	AC	Column (τ=26 min)	Raw	8.3	-	-	-	40	-	-	-	-	-	-	65	-	-	-	GAC

*AC = Activated Carbon Adsorption (GAC = granular activated carbon; PAC = powdered activated carbon)
IX = Ion Exchange

TABLE B-9. Bench-Scale Experimental Data for Alkali and Alkaline Earth Metal Removal for the Physical Treatment Processes.

Reference	Process*	Test	Leachate Type	pH	Influent Concentration, mg/l					Removal, %					Comments
					Ca	Mg	Mn	K	Na	Ca	Mg	Mn	K	Na	
151	AC	Batch (8 g/l)	Raw	-	178	100	25	380	-	42	20	87	3	-	*PAC
242,243	AC	Continuous (1 l/min)	Raw	7.1	152	132	7.2	280	374	0	0	21	0	0	GAC (6-14 mesh)
	IX	Continuous (1 l/min)	Raw	7.5	181	164	6.1	364	585	22	26	48	62	0	Glauconitic sand
207	IX(+)	Batch (2 g/l)	Biologically Treated Effluent	7.5	30	18	-	100	250	30	75	-	20	10	Dowex 50W H^+; τ = 1 hr
	IX(+)	Batch (4 g/l)	Biologically Treated Effluent	7.0	30	18	-	100	250	75	99	-	80	70	Dowex 50W H^+; τ = 1 hr
	IX(+)	Batch (12 g/l)	Biologically Treated Effluent	3	30	18	-	100	250	90	99	-	90	90	Dowex 50W H^+; τ = 1 hr
	IX(Mixed Resin)	Batch (2 g/l)	Biologically Treated Effluent	7.5	15	15	-	65	200	80	95	-	10	30	Dowex 50W H^+; Dowex 10 H^-; τ =1 hr
	IX(Mixed Resin)	Batch (4 g/l)	Biologically Treated Effluent	6	15	15	-	65	200	95	95	-	50	85	Dowex 50W H^+; Dowex 10 H^-; τ =1 hr
	IX(Mixed Resin)	Batch (12 g/l)	Biologically Treated Effluent	5.5	15	15	-	65	200	-	-	-	95	99	Dowex 50W H^+; Dowex 10 H^-; τ =1 hr

* AC = Activated Carbon Adsorption (GAC = granular activated carbon; PAC = powdered activated carbon)
IX = Ion Exchange; (+) = cationic exchange resin